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AN ASSESSMENT OF IMPACTS OF LANDFILL COMPOSITION ON SOIL QUALITY, HEAVY METAL AND PLANT HEALTH: A CASE OF LUMBERSTEWART LANDFILL IN BULAWAYO, ZIMBABWE

by

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DEDICATION

I dedicate this study to a very special person who nurtured and mentored me throughout my academic and personal life.

DECLARATION

I Peace Makuleke hereby declare that the work presented in this study for the Master of Science Degree in Environmental Science is my original work. It has never been submitted to any university or other institution for any qualification. This work has never been presented by any persons whether written or in the form of photos, graphs charts or any information. All sources quoted are recognized by way of inclusion in a comprehensive list of references at the end of the report.

Author's signature: P.Makuleke Date: 28 February 2019

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ABSTRACT

Landfills have served as the major sites for waste disposal in both developed and developing countries. Upon closure of a landfill site, the surface could be converted to a golf course, recreation park, playground, animal refuge, tennis court and industrial site. Even when closed, landfills still have the potential to contaminate the surrounding environment as a result of the migration of leachate from decomposing waste contained in the site. This study focused on assessing the impacts of a closed landfill on soils and plants at Lumberstewart closed landfill site in Bulawayo, Zimbabwe. Soil samples were collected at three different depths (0-30 cm, 30 - 60 cm and 60-90 cm) at the landfill and a control site. The soil samples were analysed for their texture, pH, electrical conductivity, organic matter content, cation exchange capacity and concentrations of Cd, Cu, Cr, Fe, Ni and Zn. Samples of jimson weed and pigweed growing at the closed landfill and the control site were collected from the same sites where soil samples were collected, and the concentrations of the same set of heavy metals in these weeds determined. Soil samples were digested using EPA method 3050B: Acid Digestion of Sediments, Sludge and soils whereas nitric acid and hydrogen peroxide was used for digestion of plant samples. Both plant and soil digests were analyzed for heavy metals concentrations using Flame Atomic Absorption Spectrometry (AAS). Soils from the landfill as well as the control site had a high content of sand with soil pH values which were alkaline. The electrical conductivity values of the soil samples were relatively low ranging from 0.39 to 1.67 dS/m, indicating low levels of salts in soils at the landfill. The concentrations of heavy metals at the closed landfill site were higher than the control site. Heavy metals concentrations in soils at the closed landfill followed the order Fe>Zn>Cu>Cr>Ni>Cd. Results indicated that Fe was exceptionally higher than the other metals with concentration values averaging 45690±17255 mg/kg. Cadmium on the other hand had the least concentration with values of 0.01±0.00 mg/kg. Values of Enrichment Factors of heavy metals around the soil at different depths indicated that the enrichment of heavy metals increased with depth at the landfill up to 30-60 cm after which a decrease was observed. Values for heavy metal Contamination Factor of soils around the landfill ranged from low concentration (CF<1) to very high concentration (CF>6). The Pollution Load Index (PLI) values for the soil at the Lumberstewart landfill indicated that all sites were polluted (PLI>1). Site 6 had significantly higher mean concentration of heavy metals in soils at the landfill whereas site 11 had the least. The concentrations of Cd and Ni in soils at the landfill were below

permissible limits of South African National Norms and Standards (NNS) as prescribed by NEMA (2008) in South Africa whereas Cr, Cu and Zn in soils were above the NNS permissible limits. Heavy metal concentrations in soils at the landfill were above World Health (WHO) permissible limits except for Cd which was equal (0.01 mg/kg) to the permissible values of Cd in the soils at sites 5, 8, 9, 10, 11 and 12. Mean concentrations of heavy metals in jimson weed and pigweed were in the order Fe>Zn>Cu>Cr>Ni>Cd. The concentrations of Cd, Cr, Cu, Fe and Zn in both plants from all sites at the landfill were significantly higher than the control site. Heavy metal transfer coefficient for both plants indicated that heavy metal uptake was more species dependent than soil heavy metal concentration dependent. The results from this research indicate that though the Lumberstewart Landfill has been closed, it is still affecting the soils in the vicinity of the landfill. Plants and water around the Lumberstewart closed landfill could be at risk from heavy metal contamination. High concentrations of heavy metals observed in the soil could present a health risk to communities should they decide to use the landfill site for arable purposes.

Keywords: *Landfill, heavy metals, leachate composition, contamination factor, transfer factor, Pollution Load Index, Pigweed (Amaranthus), Jimson weed (Datura Stramonium).*

TABLE OF CONTENTS

Title **Page No**.

LIST OF FIGURES

LIST OF TABLES

LIST OF APPENDICES

LIST OF ABBREVIATIONS

BOD Biochemical Oxygen Demand

COD	Chemical Oxygen Demand
EC	Electrical conductivity
EMA	Environmental Management Agency
TOC	Total Organic Carbon
UNEP	United Nations Environment Programme
USEPA	United States Environmental Protection Agency
WHO	World Health Organization

CHAPTER 1
INTRODUCTION

1.1 Background

Municipalities and industries are the major generators of waste worldwide and landfills have been used as waste disposal sites by these institutions for many years. According to Salami et al. (2013), municipal waste is mainly disposed of in landfills worldwide as they offer the opportunity of disposing huge volumes of municipal solid waste and are cheaper compared to other disposal methods like incineration. Landfilling is the most effective waste disposal method used in Zimbabwe like in many other countries. Unlike an open dumpsite, the design of a landfill undergoes thorough planning and management from site location to closure of the landfill to ensure that waste disposed in the landfill does not contribute to environmental pollution.

Wastes disposed in landfills undergo decomposition gradually, and rainwater could infiltrate and percolate through the waste layers in a landfill, removing soluble compounds encountered in the refuse (Jeong-Hoon et al., 2001). The liquid generated during waste decomposition processes and rainwater infiltrating the landfill (both of which are percolating the waste) is referred to as landfill leachate. According to Mahavidyalaya and Ward (2012), leachate picks up a variety of suspended and dissolved materials as it further moves downward through waste disposed in the landfill. The composition of leachate from landfill is affected by many factors including prevailing temperature, moisture content of waste and availability of oxygen, but mainly the waste type disposed on the particular landfill as well as its age. Different reactions in the landfill occur during different periods of the landfill life. For example, young landfills produce leachate having high biological oxygen demand (BOD) and chemical oxygen demand (COD) which gradually decline in the first 10 years. The content of organic compounds in the landfill are reduced at a higher rate compared to inorganic ones as the landfill ages because of the biodegradable nature of organic compounds/materials (Adhikari et al., 2014).

Landfill leachate is the greatest contamination threat to soils and plants around a landfill environment since it contains toxic substances, especially if the waste is from the industrial sector and the landfill is not lined. The landfill leachate infiltrates and percolates through soil pores and

could contaminate surface and the subsurface environments (Gaur and Adholeya, 2004). The properties of soils around the landfill determine the rate at which landfill leachate infiltrates and percolates into the ground. Soil texture, porosity and soil pH affect the movement of pollutants within the leachate plume that infiltrate into the subsurface (La Bauve et al., 1988; Sherene, 2010; Harter, 1983). Soils like silt and clay which have fine texture are characterized by low permeability as their small pores can clog easily leading to slower movement of leachate within the soil environment and a reduction in contaminant adsorption and filtration. According to Vennum and Nejedly (1990), fine textured soils have high concentrations of contaminants such as heavy metals because of their high sorption capacity. In contrast, coarse textured soils facilitate infiltration of leachate at the expense of filtering out contaminants due to large pores in these soils. High soil pH decreases the mobility of contaminants especially heavy metals which form precipitates at alkaline pH levels whereas a decrease in soil pH facilitates the movement of contaminants because of increased solubility at acidic pH.

According to Tanee and Eshdomi-Mario (2015), the decay of solid wastes releases substances into the leachate that can increase the concentration of heavy metals in the surrounding soil and alter the natural balance of nutrients available for plant growth and development. Kanmani and Gadhimath (2013) argue that, non-hazardous waste such as incinerator ashes, concrete, glass, plumbing fixtures and window frames disposed at a landfill also produce heavy metals that could pollute the environment. Several researches have shown that there are increased levels of heavy metals notably chromium (Cr), cadmium (Cd), iron (Fe), copper (Cu), nickel (Ni) and zinc (Zn) in soils around dumpsites (Tanee and Eshdoni-Mario, 2015, Salami, et al., 2013, Gaur and Adholeya, 2004) highlighting landfills as sources of heavy metals to surrounding environments. Elements and compounds contained in the leachate spread into the surrounding environment through processes such as infiltration, percolation and lateral movement. This may affect species diversity and agricultural productivity of soils in the vicinity of the landfill. Teta and Hikwa (2017) pointed out that heavy metals have toxic effects on the environment and on health and this has been of great concern for many years. They accumulate in plants and animal tissues causing toxicity. Metals such as Cd and Cr are linked to kidney disorders, brain damage and anomalies in metabolic functions of these organs in man (Teta and Hikwa 2017).

The closure of a landfill refers to cessation of disposal activities at the site. The landfill site no longer receives waste and is prepared for post-closure maintenance using an approved plan. According to Teta and Hikwa (2017), a landfill is closed in accordance with the applicable regulations, statues and local ordinances in effect at that given time in the country. However, even when a landfill is closed, leachate continues to be produced. Monitoring and maintaining the landfill for a period of at least 30 years has been recommended to decrease the negative impacts of leachate on the environment around closed landfills (Shemdoe, 2010). Leachate migration can be a serious issue at closed landfill sites. It degrades resources such as water, soils and plants posing a threat to health and safety of people living near the site. Most popular uses for closed landfills include golf courses, recreation parks, play grounds, animal refuge, tennis court and industrial site (Kihamba et al., 2011; Shemdoe, 2010; Cortez and Ching, 2014). Soils around landfills are usually fertile and therefore good for growing vegetables and fruits (Salami, et al., 2014; Shemdoe, 2010). Researches indicate that vegetables and food crops grown around closed landfills are likely to accumulate high levels of contaminants from soils contaminated by leachate from landfills (Garcia et al., 2007; Xiong, 1998; Cobb et al., 2000).

Pigweed (*Amaranthus)* and jimson weed (*Datura stramonium*) are dominant plants found in Zimbabwean landfills. These plants are generally found in gardens, abandoned fields and waste lands (Teta and Hikwa, 2017, Oluremi et al. 2013). According to Martiroysan (2007), pigweed is used for various medicinal purposes for instance, the plant seeds are used to make oils used to cure cardiovascular disease. Jimson weed is characterized by long, thick, fibrous roots and the stem is thick, smooth and pale yellow green to reddish purple in color. The seeds are covered by spines and are egg-shaped. Like pigweed, jimson weed seeds and leaves are used for medicinal purposes in some communities in Zimbabwe (Maranda and Bhat, 2010; Alegbejo, 2010; Kadoshnikov et al., 2005). Growing of pigweed and jimson weed in polluted soils might cause accumulation of heavy metals in plants which could compromise their potential use by the local community. It may also result in health problems if consumed by humans (Oluremi et al., 2013).

1.2 Research Problem

Zimbabwe like many other developing countries has a challenge of managing waste impacts on the environment. Landfilling is a common practice of waste disposal in all cities of Zimbabwe and many of the landfills are un-monitored. Some of the landfills are also unlined. In unlined landfills, leachate generated in the landfill may be infiltrating into ground water and soils causing severe contamination. Lumberstewart landfill was in use by 1989 when there were no regulations governing the design and use of landfills in Zimbabwe. The closed landfill was constructed without any liners at the bottom to impede landfill leachate movement into the environment. The base was simply compacted to minimise seepage of leachate into the surrounding subsurface environment. Lack of an appropriate bottom liner facilitates the rate of movement of leachate from the landfill to surrounding environments presenting serious pollution problems. Kumar (2014) states that, due to minimum migration control mechanism at landfill sites, there is subsequent migration of leachate through the bottom of the landfill hence promoting environmental pollution. No research has been done to establish whether leachate from Lumberstewart landfill is migrating into surrounding environments. This study will provide more information in this regard.

At Lumberstewart landfill, though much of the waste deposited in the landfill comprised of organic material, considerable amounts of plastic, batteries and metal rubbish which release heavy metals like Cd, Cr, Cu, Pb, Ni, Fe and Zn were also deposited in the landfill site. These heavy metals could be contained in the leachate generated form the landfill. According to Karatas et al. (2006), heavy metals are non-biodegradable and have the ability to accumulate in soils and this is dangerous to animals and plants. Plants absorb heavy metals through various processes and these metals could easily enter the food chain in the ecosystem. Heavy metals entering the food chain in high concentrations can lead to severe health problems in animals and plants. High concentrations of heavy metals can hinder the growth of plants and hinder nutrients and water uptake (Shemdoe, 2010; Kihamba et al., 2011) which could result in damage of root tips and enzymes (Sekara et al., 2005). According to Kihamba et al. (2011) heavy metals can damage mental and nervous function of animals, lowering energy levels and damaging lungs, kidneys and the liver. It is imperative that studies which establish the concentrations of heavy metals in soils around the landfill be determined to expose potential risk to animals and plants.

The build-up of contaminants in soils such as heavy metals and their uptake in the vicinity of Lumberstewart landfill by plants is of great concern. Heavy metals might enter the human body through consumption of polluted water or ingestion of crops grown on polluted land (Dudka and Miller, 1999; Cambra et al., 1999). The Lumberstewart landfill is currently being used for agricultural activities. It is not known whether these plants are absorbing trace metals from soils around the landfill. Uptake of heavy metals by the plants may affect yields and expose consumers to heavy metals with associated consequences. Continuous uptake of heavy metals by pigweed and jimson weed growing in the vicinity of the landfill could affect the quality of seeds produced by these plants which are used for making oil and cereals. According to Sekara et al. (2005), high amount of heavy metals in pigweed and jimson weed can lead to reduction of protein and fibre content and vitamins in the seeds, affecting the diet of human beings and animals consuming these plants.

Leachate causes ground water pollution and this result from rainwater which passes through the waste in the landfill. A closed landfill may continue producing leachate with contaminants if not capped properly with impermeable material before the closure of the landfill. Leachate infiltrates and percolates through pores into ground water aquifers which cause pollution. Ground water is abstracted for various activities such as agricultural, domestic and industrial activities. According to Al Raisi et al. (2014), ground water around the landfill is sometimes used for watering crops and domestic activities such as cooking and drinking. Consuming water from a contaminated environment may cause heavy metal poisoning to users. The contaminated water if used for bathing and washing might cause allergies and skin infections (Mangore and Taigberu, 2004, Maranda and Bharat, 2010). Polluted ground water results in poor quality drinking water and reduced water supplies in communities around the landfill which depend on groundwater resources for various purposes. Contaminated ground water requires attention since this is a source of water for many populations. Soils serve as a water purification system. Its contamination by heavy metals would imply potential water resources in the vicinity being polluted with same. This study will determine the concentration of selected heavy metals in soils and plants around the landfill, to understand the impact of the landfill on the heavy metal concentration in the surrounding environment and create awareness concerning pollution from landfilling of waste.

1.3 Research Aim

The aim of this study is to assess the extent of heavy metals contamination in soils and plants around a closed landfill site in Bulawayo, Zimbabwe.

1.3.1 Research objectives

The specific objectives of this study were:

1. To characterize soils around the Lumberstewart landfill to understand the role that they may play in leachate migration from the landfill.
2. To determine the concentration of selected heavy metals including Cd, Cr, Cu Ni Fe and Zn at different soil depths around a closed landfill.
3. To determine the effects of heavy metals concentration in tissue of pig weed and jimson weed around the closed landfill.

1.4 Research Questions

The following research questions guided the research:

1. What are the concentrations of the selected heavy metals in soils around the closed landfill?
2. How do these heavy metal concentrations in soils around the closed landfill vary with depth?
3. What is the amount of these heavy metals in pigweed and jimson weed around the closed landfill?

1.5 Justification of the Study

In the past decades, Zimbabwe's population growth has increased at an alarming rate especially in cities and towns. This population growth has contributed to vast amounts of waste generated and disposed in municipal solid waste landfills. Landfilling is a common waste disposal practice in all Zimbabwean cities and many are unsecured, unmonitored and easily accessed. A lot of research has been done around the impact of landfill leachate on quality of surface and ground water (Mangizvo, 2008; Mangore and Taigberu, 2004: Kubare et al., 2010). The findings of the studies indicate that surface and ground water near the landfills are contaminated but there is lack in-depth

of the study which investigates the effect of a closed landfill on soils and plants in Bulawayo. In Zimbabwe, data on heavy metals in soils around closed landfill is also lacking. Risks presented by a closed landfill include gas migration and soils, plants and ground water contamination. These risks might cause a threat to the health and safety in the communities near the area. Therefore, an important area of study is to assess the extent of heavy metal contamination in soils and plants in the vicinity of a closed landfill.

Heavy metals such as Cd, Cr, Zn, Fe, Cu and Ni enter the human body through consumption of drinking water and food crops, containing the metals and inhalation (Dudka and Miller, 1999). The chemicals are absorbed into the body where they are transported to the liver through bloodstream. The consumption of heavy metals by humans through food has been reported by several scholars in many countries (Islam et al., 2007; Salami et al., 2013). Continued exposure to heavy metals like Cr damages body organs such as liver, kidney and nerve tissues.

The results of this study are expected to help understand the extent of pollution at a closed landfill site by detailing how concentration of heavy metals in soils and plants deviate from acceptable standards. This information will assist in the design of management strategies that would be effective in remediating the contaminated environment. The determination of the extent of heavy metal uptake by plants around the landfill will reveal whether food crops grown in the closed landfill are safe for human and animal consumption. Determining the variation of heavy metal contamination with depth around the landfill will provide an indication of whether ground water in the vicinity is at risk of contamination.

The results of this study will therefore ascertain whether it is necessary to put in place solutions to prevent groundwater contamination in the region or not. Information on the heavy metal uptake by pigweed and jimson weed is relevant to the community for an appreciation by the community of how these plants have accumulated heavy metals and the potential health risks they may be exposed to by consuming these plants. It is also expected that the findings of the study on heavy metal concentration in different species will widen knowledge on species which accumulate high amount of heavy metals than the others in the environment. This will provide basis for measures to be implemented on the closed landfill.

1.6 Outline of the Dissertation

Chapter 1: INTRODUCTION- Provides the background to the study. It also provides the research problem, research aim and objectives, research questions and justification of the study.

Chapter 2: LITERATURE REVIEW- Focuses on the description of the landfill, landfill design, and legislation related to management of landfill operations in Zimbabwe, leachate generation and composition, factors affecting landfill leachate composition and closure of landfills. It also provides information on heavy metals contamination around landfill environments, uptake of heavy metals by plants from leachate contaminated soil and effects of heavy metals from landfills.

Chapter 3: RESEARCH DESIGN AND RESEARCH METHODS- Outlines a brief overview of the area under study, ethical clearance, soil and plant sampling and quality assurance.

Chapter 4: RESULTS AND DISCUSSION-Focuses on the results of the study and discussions. Results are presented in tables and graphs

Chapter 5: SUMMARY OF FINDINGS, CONCLUSION AND RECOMMENDATIONS- Provides the summary of findings and conclusion of the study. It summarises recommendations based on the findings obtained from the study.

CHAPTER 2

LITERATURE REVIEW

2.1 Introduction

This chapter reviews the literature that informed the study on assessment of the impacts of a closed landfill on soils and plants.

2.2 Description of a Landfill

Vast amounts of wastes from urban municipal and industrial sectors are generated worldwide and for several decades landfills have been the ultimate disposal sites for all types of wastes including residential, commercial and industrial wastes (Farquhar, 1989). According to Mohammad (2010), one of the affordable means of disposing waste in the world is landfilling and it accommodates huge amounts of municipal waste as compared to other methods like incineration. A landfill is an area that is designed and constructed to dispose unwanted material by burial to reduce the release of contaminants that pollute the environment (Kumar, 2014). The designated space is carefully engineered for discarding unwanted material and eliminating the risk of pollution to the community. At the end of every day, the disposed waste piled in layers is compacted with the aid of heavy machinery and buried in order to reduce odors, confine the waste and reduce the amount of water percolating into the waste. When the constructed landfill is full of waste, it is covered with a thick layer of soil and can be used for various activities such as sports fields, golf courses and pasture land. Decay of disposed solid waste and percolation of rain water into the landfill results in the production of a heavily contaminated waste water called leachate. Recent landfills have liners at the bottom which reduce the migration of leachate out of the landfill. In landfills, there are chemical, physical and biological processes facilitating the decomposition of the disposed waste. Farquhar (1989) states that the main environmental challenges experienced at landfills are associated with the movement of leachates from the landfill to the surrounding underground water, surface water and soils.

2.2.1 Landfill Design

The landfill is designed and constructed in a way to reduce the flow of leachate into the soils surrounding. Liners, pumps, pipes and monitoring wells are included in the design and construction of most modern landfills to restrict and capture leachate flow. The landfill bottom liner is the most important structure when designing a landfill (Bouazza and Van Impe, 1998;

Moreno, 2011). Waste is piled and compressed on top of the bottom liner which is made of rugged resistant material to reduce movement of leachate. The cover system is another crucial part when designing the landfill since its aim is to prevent infiltration of rainwater and consequently the generation of leachate. Daily, intermediate and final covers are three types of covers used to cap landfills and these have different purposes in the landfill. According to Kamaruddin et al. (2013), daily cover thickness of about 0.6 m prevents animals from spreading disposed waste from the landfill to other places nearby, which if not prevented may result in spreading of diseases and pollution of the environment. The intermediate cover which has a thickness of about 1.2 m reduces the amount of rainfall infiltration into the waste. The final cover could be 2 m thick and its function is to promote runoff of rainwater and prevent infiltration into the waste, thereby reducing leachate production. Figure 2.1 shows a cross section of the landfill structure.

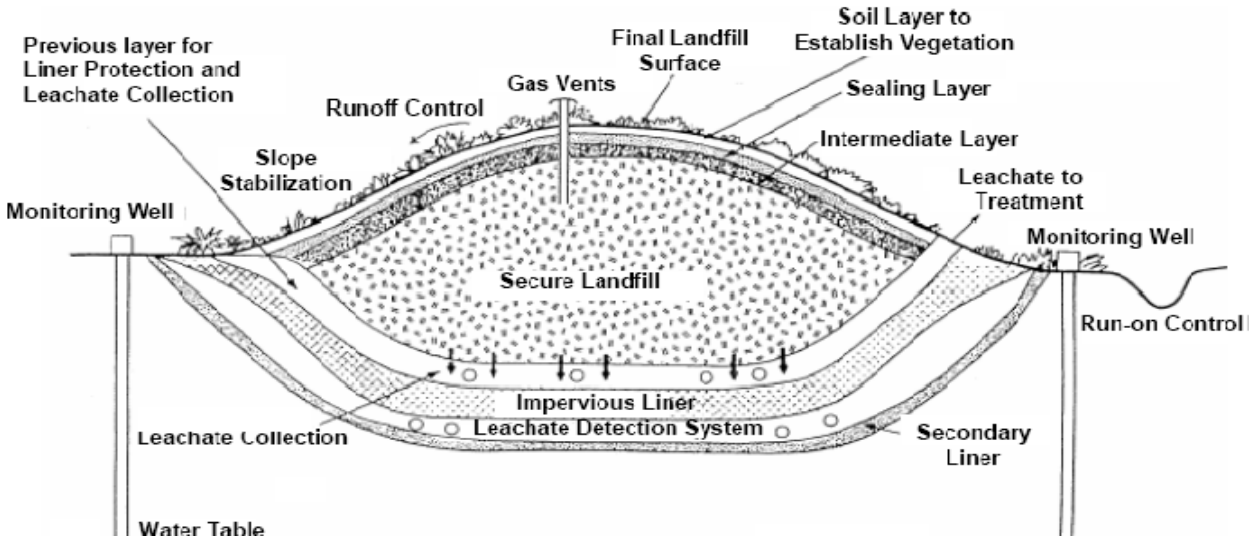


Figure 2.1: Cross section showing the structure of a landfill (Haubrichs, 2006)

2.3 Legislation related to management of landfill operations in Zimbabwe

As a result of the negative impacts of landfills on surrounding environments, several countries have legislations and guidelines on the location and designing of landfills. In Botswana for example, Waste Management Act 65:05 state that landfills should not be located within 500 m of surface water used for domestic or human consumption. Cover material used for capping the

landfill includes ash, soils and builder's rubble. In Swaziland, a new landfill site should not be located less than 300 m from an old landfill according to the Environmental Management Act, No. 5 of 2002. These regulations are similar to those in developed countries like USA and Australia. In Zimbabwe, the operations of landfills are guided and managed by the Environmental Management Agency (EMA), using the Environmental Management Act as the major controlling tool. The 2007 regulations (CAP 20:27) defines precisely all the requirements that the operations are supposed to follow. The regulations are listed under the Statutory Instrument 6 of 2007, Part 1V, and Section 22. Under Section 22:2, it is stated that all new waste sites are supposed to have proper liners designed according to the type of the environment where the landfill is located and this is approved by EMA. The act states that all new solid waste sites shall be lined with appropriate lining specific to the type of the environmental risk. Landfill sites in Zimbabwe must be 60 m from lakes and streams, 1500 m from human habitation and 2000 m from airport as stipulated by regulations (Teta and Hikwa, 2017). After rehabilitation of a closed landfill site, the area can be used for parks and golf course. There is limited information in Zimbabwe on rules to do with landfills for example information on cover material used at the landfill and type of activities done on closed landfills.

2.4 Landfill Leachate Generation and Composition

In landfills generally, micro-organisms are responsible for gradual decomposition of solid wastes. Rain water infiltrates and percolates through the cover of the landfill, washing down soluble compounds in the landfill produced by chemical and biological processes taking place within the landfilled waste. The movement of water through the landfill removes soluble compounds from the waste leading to the formation of leachate. Kamaruddin et al. (2013) states that leachate from a landfill usually has four major classes of pollutants and these are dissolved organic matter, inorganic macro components (calcium, sodium, chloride, and iron), heavy metals (cadmium, copper, zinc, chromium etc.) and xenobiotic compounds (pesticides, hydrocarbons etc). According to Kjeldsen et al. (2002), dissolved organic matter that is in leachate includes the organic decomposition products like volatile acids, refractory fluvic and also humic-like compounds. The dissolved organic matter components of landfill leachate include chemical oxygen demand (COD), total organic carbon (TOC), biological oxygen demand (BOD) and organic nitrogen. The presence

of decomposed organic matter in the waste influences the color of leachate produced, with common colors of leachate being yellow, brown and black (Aziz et al., 2007). The major inorganic macro-components of landfill leachate include Na, Mg, Ca, K and Fe. Kjeldsen et al. (2002) argue that the level of macro-component concentrations in leachate is influenced by stabilization processes in the landfill. Table 2.1 shows a typical composition of landfill leachate.

Table 2.1: Typical composition of landfill leachate (mg/L)

Parameter	**Range**
pH	4.5-9.0
Spec. Cond. (µS/ cm)	2500-35 000
Total solids	2000-60 000
Organic Matter	
TOC	20-57 000
BOD	140-152 000
COD	0.02-0.80
BOD5/COD (ratio)	14-2500
Organic nitrogen	30-29 000
Inorganic Macro-components	
Total phosphorus	0.1-23
Cl	150-4500
SO^{2-}_4	8-7750
HCO^3	610-7320
Na	70-7700
K	50-3700

Parameter	**Range (mg/L)**
$N_2H_4O_3$	50-2200
Ca	10-7200
Mg	30-15 000
Heavy metals	
As	0.01-1
Cd	0.0001-0.4
Co	0.005-1.5
Cr	0.02-1.5
Cu	0.005-10
Pb	0.001-5
Hg	0.00005-0.1
Ni	0.015-3
Fe	3-5500
Zn	0.03-1000
Mn	0.03 – 1400
Si	4 – 70

Source: Kjeldsen et al., 2002

Heavy metals are considered as one of the major components and essential parameters of leachate characterization. Common heavy metals identified in landfill leachate include chromium (Cr), (cadmium), zinc (Zn), lead (Pb), nickel (Ni) and copper (Cu) among others. Heavy metals contained in municipal solid waste may end up in surrounding soils and move downward facilitated by percolation of rain water (Alloway and Aryes, 1997; Oyedele, 2008). Continuous

monitoring of contaminants in landfill leachate has been done as recommended by authorities such as the city councils in most countries and are normally managed by landfill operators (Kjeldsen et al., 2002). According to Bjerg et al. (2003) generally, heavy metal concentrations in landfill leachate originating from municipal solid waste landfill are low and rarely lead to ground water pollution around the landfill. However, if present in high concentrations as is the case with leachates from hazardous or mixed municipal and hazardous waste landfills, metals such as Zn, Cr, Cd, Pb and iron (Fe) can be toxic (Sawaittayothin and Polprasert, 2007). In addition, some household and industrial chemicals produce xenobiotic compounds which at high concentrations may also have negative environmental impacts. According to Kjeldsen and Christensen (2000), the generation of xenobiotic organic compounds decrease with age of landfill but this depends on the rate of decomposition and fermentation processes that go on in the landfill.

2.5 Factors Affecting Landfill Leachate Composition

There are various factors influencing the composition of landfill leachate including type of wastes disposed, climate of the area and landfill management.

2.5.1 Climate

The quantity and quality of leachate produced in landfills is dependent mainly on the temperature and rainfall received in a particular area. High amount of precipitation increases potential infiltration in the landfill and consequently leachate generation. High water content dissolves and flushes out soluble inorganic and organic compounds and microbial cells from the landfill (Slomczynska and Slomczynski, 2004) leading to production of high amounts of leachate. High amount of precipitation could also cause leaching of contaminants in waste disposed in the landfill leading to lower level of pollutants in the leachate generated. The temperature influences the quality and quantity of leachate through its influence on microbial activities. High temperatures increase bacterial growth, microbial processes and chemical reactions within the waste in the landfill and a consequent increase in the rate of biodegradation of waste. During wet and hot season, bacterial growth and chemical reactions increase due to enzyme activation caused by increase in moisture content in the landfill. The availability of moisture facilitates fermentation of organic matter and decomposition is faster during wet, hot season as compared to cool, dry season (Moreno, 2011). This could lead to large volumes of leachate containing various contaminants.

During dry cool and dry hot seasons, the amount of landfill leachate produced may be reduced due to high rates of evaporation, which reduces moisture in the landfill. However, leachate produced under these conditions is highly concentrated and therefore more toxic. According to Adhikari et al. (2014), landfills with low moisture content that is more than 20% but less than 40% of water is characterized by slow stabilization and the rate of biodegradation is slow. The decrease in the temperature reduces the growth of bacteria and chemical reactions occurring at the landfill due to deactivation of enzymes.

2.5.2 Age of the Landfill

The landfill's age also influences its leachate composition. Leachate from landfills can be divided into different categories according to age of the landfill namely young, medium and old. Recent or young landfills contain vast amounts of biodegradable organic materials. Moreno (2011) states that leachate from landfills less than five years old are in the acidogenic phase. The landfill contains huge amount of biodegradable organic matter which usually goes through anaerobic fermentation helped by water in the landfill, producing volatile fatty acids. The landfill leachate produced at this phase is acidic (Table 2.2). Landfills between 5 and 10 years are of medium age and are characterized by acidogenic and methanogenic features and those that are more than 10 years are believed to be aged and stabilized. The pH of leachate from such landfills is alkaline as compared to leachate from young landfill which is acidic. Those landfills which have existed for long produce stabilized leachate with relatively low COD and low biodegradability (Bashir et al., 2009). Table 2.2 below shows the composition of leachate at different ages of landfill.

Table 2.2: Composition of leachate at different ages of landfill

Parameter	**Young**	**Intermediate**	**Old**
Age (yrs.)	<5	5-10	>10
pH	6.5	6.5-7.5	>7.5
COD (mg/1)	>10 000	4 000-10 000	<4 000
BOD_5/COD	>0.3	0.1-0.3	<0.1
Organic compounds	80% Volatile Fat Acids (VFA)	5-30% Volatile Fat Acid + humic and fluvic acids	Humic and Fulvic acids
Heavy metals	Low-medium	Low	Low

Biodegradability	Medium	Medium	Low

Source: Adhikari and Khanal, 2015

2.5.3 Waste type

The type of waste disposed in the landfill has a bearing on the composition of leachate generated in the landfill. Various types of wastes from household, commercial, industrial, institutional, municipal services and construction debris have different types of organic and inorganic materials and in different amounts (Moreno, 2011). The organic materials are degradable materials like those found in kitchen waste whereas inorganic materials are not degradable materials like plastics, glass and metals. The higher the content of organic materials in the landfill, the more important is the biological process (Adhikhari, and Khanal, 2015). This process is also accelerated by the moisture content of the waste. Waste with high moisture content will increase both the occurrence of microbial mediated processes and the quantities of leachate generated in the landfill. In a study by Moreno (2011), large amounts of paper disposed at the landfill led to the decrease of waste decomposition process, decreasing the amount of leachate generated since paper is resistant to microbial decomposition. Hazardous wastes from residential, industrial, commercial and institutional sources contribute to toxic landfill leachate. Table 2.3 below shows the origin of waste, kind of solid wastes and types of contaminants introduced into the landfill by the waste type.

Table 2.3: Origin of waste, kind of waste and types of contaminants introduced into the landfill

Origin of waste	**Kind of waste materials**	**Types of contaminants introduced into the landfill by the waste material**
Residential	Food wastes, paper, cardboard, plastics, textiles, leather, yard wastes, wood, glass, metals, ashes, special wastes (e.g. bulky items, consumer electronics, white goods, batteries, oil, tires) and household hazardous wastes	Oven cleaners, solvents, paints, glues, tars, preservative coating and gluing wastes,

Industrial	Housekeeping wastes, packaging, food wastes, construction and demolition materials, hazardous wastes, ashes, special wastes.	Waste oils, solvents, acids, paints, glues, fluorescent tubes, caustic soda,
Construction and demolition	Wood, steel, concrete, dirt etc.	Building wastes, paints, glues, tars, solvents
Municipal services	Street sweeping, landscape and tree trimmings, general wastes from	Pesticides for lawn and garden maintenance, swimming pool maintenance chemicals
Manufacturing	Industrial process wastes, scrap materials, off-specification products, slay, tailings	Cyanides, metal sludge, solvent, oils, acids, paints, calcium fluoride sludge, methylene chloride, acetone, methanol
Agriculture	Spoiled food wastes, agricultural wastes, hazardous, wastes (e.g. pesticides)	Pesticides, livestock sewage waste, nitrates, phosphates, total dissolved salts

2.5.4 Landfill management

Landfill management affects the quality and composition of leachate produced in the landfill. According to Henry et al. (2005), sorting of waste at the landfill has numerous benefits. Prior to disposal, sorting of waste can be done as a way of preventing the disposal of hazardous waste in the municipal solid waste landfill and this influences the composition and toxicity of leachate. Fencing of landfill and controlling access also prevents unofficial disposal of waste containing metals like Cr, Cd, Cu, Pb, Fe, Ni and Zn. Poor management of landfill results in poor compaction of waste and this could allow easy percolation of water. Poor compaction of waste makes aerobic conditions possible hence promoting decomposition of materials affecting the production and composition of leachate. Furthermore, downward movement of water facilitates the dissolution of hazardous materials disposed at the landfill. Poor cover layers also allow infiltration of water into the waste promoting the generation of leachate. Aerobic decomposition of organic solid waste at the landfill produces landfill leachate and landfill gas. Landfill leachate collection systems and methane venting systems are of great importance in the management of landfill. Leachate

collection systems are made up of a drainage layer, the drainage pipes, collection pipes and shafts (Henry, 2005). Perforated pipes run throughout the landfill in order to collect landfill leachate and these pipes drain and carry leachate to the leachate collection pond. Landfill gas consists approximately 50% methane, 50% carbon dioxide and small amounts of oxygen and nitrogen. Methane gas from landfills can explode so it must be monitored and continuously removed from the landfill. To collect gas from the landfill, pipes are fixed into the surface within the landfill to tap gas and this gas is vented. Burning or venting methane gas reduces the negative effects of gas at the landfill.

2.6 Closure of Landfills

When closing a landfill, the site's specific characteristics and affordability of the waste management options are taken into consideration. According to Li (2009), there are two methods available when closing a landfill and these are evacuation and in-situ method. Evacuation refers to the closing of the dumpsite by removing the waste from the site. In this case, waste is graded into different classes for recycling purposes. In-situ method involves the closing of the landfill by covering the waste to reduce the infiltration of rainwater, access by vectors and spread of waste by wind. According to Li (2009), people and animals are prevented from scavenging the waste and continuation of waste disposal at the site is halted. This in-situ method also includes the controlling of odor and risk of veld fires at the site. During closure of landfills, clay rich material is used to cover the waste, which helps to maximize runoff at the expense of infiltration hence slowing down leachate generation. According to UNEP (2005), a final top layer can be of other soil types and has the purpose of protecting the clay layer and providing a growing space for vegetation. To prevent damage of clay layer, deep rooted plants are usually avoided on landfill site (Benson and Ebong, 2005). UNEP (2005) argues that closed landfill sites should be vegetated with short rooted plants and could preferably be used as a recreational area, open meadow or a protected habitat for wild and local species. Cover soils in landfills are regarded as fertile hence used for cultivation of various crops and a similar activity exists at the closed landfill in Bulawayo. Areas around landfill have been used for agricultural activities since the soils are fertile though various studies indicate that plants such as vegetables accumulate huge amounts of heavy metals like Zn, Cr, Cu, Pb and Fe from contaminated soils (Cobb et al., 2000; Benson and Ebong, 2005).

2.7 Heavy Metals Contamination around Landfill Environments

Heavy metals are found in ecosystems in different concentrations. Major contributors of heavy metals into landfills include industrial waste, mining waste and domestic waste. Waste such as batteries, electrical gadgets and scrap materials produce heavy metals which contaminate the environment. The most common undesirable metals encountered in landfills are Cd, Zn, Pb, Cr, Cu and Ni (Kumar, 2014; Ugwoha and Emete, 2015; Salami et al., 2013). Gworek et al. (2016) analysed the concentration of heavy metals in soils around Talajaj landfill in Polland and the following results were obtained: Pb-108.5g/mg; Cu-90g/mg; Zn-560g/mg; Cr-101.5g/mg and Cd-224g/mg. A similar research was also carried out in India at Allahabad dumpsite to assess the concentration of heavy metals in soils and the study revealed that heavy metals concentrations ranged between 32.46 ± 1.07 to 108.85 ± 3.99mg/kg. Heavy metal concentrations around the dumpsite were in the order Zn>Fe>Ni>Cu>Cr>Cd. Heavy metals were also analyzed in plants and soils at Gunung Tugel closed landfill in Banyumas-Central Java. The results obtained were as follows: 6.27-34.71mg/kg,0.17-042mg/kg, 28.29-48.69mg/kg, 18.997-32572mg/kg, 342.74-834.49mg/kg, 136.10-290.14mg/kg in soil and 0.01-170mg/kg, 0.00-0.26mg/kg, 0.79-10.46mg/kg, 13.88-61.46mg/kg, 18.79-50.56mg/kg, 87.27-273.22mg/kg in plants for Cr, Cd, Cu, Fe, Mn and Zn respectively (Mohammed, 2010). Pollution of biosphere with heavy metals induced by landfilling poses serious problems to the environment surrounding landfills. The extent to which these environments are affected is determined by the properties of the surrounding soils.

2.7.1 Soil properties influencing leachate and contaminant migration in soils around landfills

Analyses have shown that the behavior of metals in soil and their retention in the solid phase of soil is affected by pH, the quantity of the metal, cation exchange capacity, content of organic matter and mineralogy of the soil (Aydnalp and Marinova, 2003). Soil type plays an important role in heavy metal bioavailability and, therefore, toxicity of heavy metals in the soil environment (Evanko and Dzombak, 1997). The texture of the soil refers to the ratio of clay, silt and sand sized particles that make up the mineral fraction of the soil. It determines the rate at which liquids can migrate within the soil environment and therefore determine the soil's infiltration capacity and permeability. Soil permeability controls the rate at which leachate migrates within the soil environment. Fine textured soils reduce the movement of leachate in the landfill since soil particles

are small and closely packed. Leachate takes a longer time to move through fine soil particles such as clay as compared to coarse textured soils which have large particles facilitating the movement of leachate. Soils with fine particles retain heavy metals because of the large surface area and adsorption capacity as compared to soils with large particles. McGrath and Loveland (1992) observed that the retention of heavy metals such as Cr, Ni and Zn in top soil increased with the increment of clay size fraction. Clayey soils have a higher CEC (Cation Exchange Capacity), which also increases the adsorption of metals. Aydnalp and Marinova (2003) reported that the fine textured soils contain higher amounts of heavy metals while coarse textured soils contain lower concentrations. Clayey soil acts as adsorbent due to its impermeability and high cation absorption capacity. According to Evanko and Dzombak (1997), the adhering nature of clay particles reduce migration of metals in the soil while the large particles of sandy soils help the metals to move faster. Clay soils reduce the rate at which metals migrate from the landfill, hence they are widely used as liners in landfills.

The movement of heavy metals in the soil environment is also affected by organic matter content. High organic matter retains metals and acts as a sink for heavy metals reducing their rate of movement. This however depends on the specific metal as the chelates and complexes formed between organic matter and some heavy metals render them more soluble and mobile in the environment. Sherene (2010) reported that organic matter is an important soil characteristic affecting the adsorption of heavy metals. Humic materials in organic matter can effectively slow down the mobility of heavy metals due to the high density of functional groups which tend to form complexes and chelates with the metals, rendering them less mobile (LaBauve et al., 1988). High organic matter content in the soils tends to have high CEC which increases the soil's capacity to immobilize heavy metals in the soil environment. According to Sherene (2010) organic matter and hydrous ferric oxide decrease heavy metal availability through immobilization of these metals.

The migration of heavy metals in the soil is also determined by soil pH. According to Kumar (2014), many metals can dissolve in acidic soil conditions compared to basic and neutral soils. Evanko and Dombak (1997) suggest that Ni, Cu and Zn have high mobility under acidic conditions. LaBauve et al. (1988), Agyarko et al., (2010) and Alloway and Ayres, 1997 revealed that the pH of the soil solution maintained at neutral and alkaline levels retards mobility of all

heavy metals due to the precipitation of hydroxides, carbonates or formation of insoluble organic complexes). This has also been reported by Kanmani and Gandimathi (2013) and Harter (1983) who indicated that a high rate of leaching of heavy metals under acidic conditions during the degradation process results in high metal concentrations in landfill leachate. With increasing pH, content of organic matter and clay, the solubility of most metals decreases due to their increased adsorption and precipitation (Kumar, 2014; Evanko and Dzombak, 1997). Where possible, it is therefore beneficial to make efforts to increase the pH of soil around landfills in order to reduce the mobility of the metals in the soil environment. Age of the landfill also determines the spread of leachate in a landfill. Heavy metals in younger landfills (0-5years) are likely to spread faster because of acidogenic phase of the leachate though contaminants have not been dissolved whereas in old landfills, heavy metals spread could be slower because of the prevailing alkaline pH conditions though much of the metals in the waste have been dissolved.

Primary and secondary minerals contained in the soil determine the CEC and mobility of heavy metals in the soil environment. Primary minerals are those which are formed from the cooling of magma and include quartz, mica and feldspar while secondary minerals in the soil are formed from the weathering of primary minerals. They are caused by transformation or chemical weathering of the primary minerals for example iron and aluminum oxides, clay minerals, dolomite, gibbsite and calcite. Primary minerals have relatively larger pore spaces and lower surface areas compared to secondary minerals and therefore have a low CEC, leading to faster movement of leachate in soils where they are dominant. Comparatively, secondary minerals such as those contained in the clay fraction of soils are negatively charged and particles have high CEC. The movement of leachate is low since particles are closely packed and can retain water and adsorb contaminants. According to Adhikari and Khanal (2015) negative charges found in soil influences the soil's ability to retain cations leading to soil cation capacity.

2.8 Uptake of Heavy Metals by Plants from Leachate Contaminated Soil

Uptake of heavy metals by plants varies with the specific heavy metal, the type of soil on which the plant is grown and the specific plant. Plant types differ in their metal uptake ability at a given site. Common heavy metals taken by plants are Cd, Pb, Cu, Cr, Fe and Zn. Some plants have the

ability to take up heavy metals to accumulate them without being destroyed by the heavy metals (Weggler et al., 2004) and are commonly used in the remediation of heavy metal polluted soils. These plant species have developed tolerance towards metals and are characterized by their ability to accumulate high quantities of metals in their tissues (Siwela et al., 2009; Oyedele et al., 2008; Intawongse and Dean, 2007). Intawongse and Dean (2007) examined the uptake of heavy metals by vegetables grown on contaminated soils and they observed that vegetables such as spinach, cabbage, lettuce, carrots and rape accumulated a high content of Zn, Cu and Mn. Heavy metals can also accumulate in cereals such as wheat, sorghum, maize and millet but at a lower rate as compared to vegetables. Sulyman et al. (2015) reported a low accumulation of heavy metals in cereals as compared to vegetables. Siwela, et al. (2009) reported significant trace metals translocation between the plant parts and the soil and observed that the translocation factor was above 1 (TF>1) implying high rate of metal uptake.

2.9 Effect of Heavy Metals from Landfills on Surrounding Soils and Plants

2.9.1 Effects on plants

Plants growing in the vicinity of landfills and other waste disposal sites could be contaminated by pollutants present in the migrating leachate. Although heavy metals such as Fe are essential for plant growth, excessive amounts can negatively affect the plant. Heavy metals absorbed in the plant cannot be easily broken down and so when the amount taken up exceeds the required quantity, negative effects could be experienced. According to Xu and Shi (2000), heavy metals hinder the absorption and transportation of important elements in plants, disturbing the metabolism and growth of the plant. Teta and Hikwa (2017) reported poor growth and depressed photosynthesis in pigweed and jimson weed at Richmond landfill in Zimbabwe caused by heavy metals. Kibra (2008) reported a notably reduction in height of rice plants growing on soils polluted with heavy metals from a landfill highlighting the adverse impacts of heavy metals on plant growth. Another study examined the impacts of heavy metals on *Lythrum salicana* and found that high accumulation of Pb and Cu resulted in rapid and complete death of the leaves and stem of the plants (Nicholls and Mal, 2003). Xu and Shi (2000) also revealed that high concentrations of Cr in *Hydrochairs dubia* prevented the plant from absorbing water. Photosynthesis is also sensitive to heavy metals toxicity and long-term exposure of plants to metals contributes to decreased leaf

growth and photosynthesis pigments in some plants. In this case the heavy metals exert their toxic action by destroying the chloroplast disturbing the process of photosynthesis. Nicholls and Mal (2003) observed that heavy metals such as Cd, Fe, Zn, Ni and Cr depress plant photosynthesis which reduces the productivity.

2.9.2 Effects of heavy metals on soils

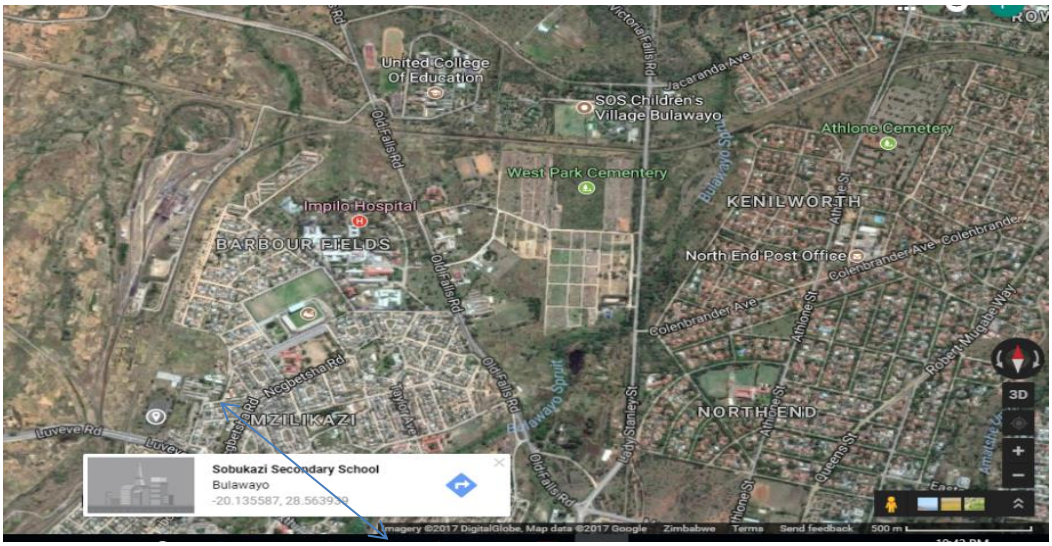
The availability of harmful metals in the soils may seriously hinder decomposition of organic contaminants (Abosede, 2017). Increased concentration of heavy metals in the soil affects microbial population activities, hence affecting the decomposition in the soil. According to Kihamba et al. (2011), heavy metals in soil slow down the speed of growth and reproduction of micro-organisms which negatively affect the functioning of an ecosystem as a whole. Research by Wu and Lin, (2003), and Ugwoba and Emete, (2015) indicate that long term soil heavy metal pollution reduces microbial metabolic activities including respiration. Some metals also compete for soil exchange sites with plant nutrients resulting in the leaching of plant nutrients, which could affect soil fertility negatively.

CHAPTER 3
RESEARCH DESIGN AND RESEARCH METHODS

3.1 Description of Study Area

Lumberstewart closed landfill is located in the western region of Bulawayo City in Zimbabwe at latitude 20° 13᾽ 55᾽᾽ S and longitude 28° 56᾽ 39᾽᾽ E (Figure 3.1). The area of study has an altitude of 1 358m above sea level. Fig 3.2 and 3.3 show the landfill site on the northern part and southern part. Bulawayo city has a subtropical climate due to its high altitude. Generally, the area has three distinctive seasons - dry, cool winter ranging from May to August; hot, dry early summer from late August to early November and lastly warm, wet late summer from early November to April. The mean annual temperature of the area is 19.6º C and October is usually the hottest month with

temperatures of over 40^{o}C being observed while coolest month is July with minimum temperatures averaging 7^{o}C (Mangore and Taigbenu, 2004). On average, the warmest month is October, coolest month is July, wettest month is December and driest month is August. Bulawayo city receives low and erratic precipitation with average annual rainfall of about 600 mm (Gariwe et al., 2017).



Lumberstewart landfill site

Figure 3.1 Location of the study area

Figure 3.2: Landfill site showing covered landfill on the northern part

Figure 3.3: Landfill site showing covered landfill on the southern part

The dominant rock type at the study area is granite rock containing quartz, feldspar and mica as the dominant primary minerals (Kubare et al., 2010; Rusinga and Taigbenu, 2005). The soils at the study area are coarse grained in terms of texture and can be classified as sandy soils. According to Chuma et al. (2013), the soil type of the region is closely related to the underlying lithology with greyish to reddish brown shallow to moderately deep soils that are associated with granite and allied rocks. According to Gariwe et al (2017), soil pH around the landfill is acidic, with low clay and organic matter contents. Lumberstewart landfill lies on a plain highveld in Zimbabwe. The area is typically covered by savanna woodland and the dominant vegetation is the acacia woodland. The closed landfill is located along an aquifer called Matsheumhlope (Kubare et al., 2010), which might have negative impacts on the quality of water being used in the city. Due to water scarcity in the region, the water table is very low averaging 15m deep because of continuous water abstraction.

The landfill site covers an area measuring approximately 300 m x 200 m. The landfill was not lined but had compacted clay at the base to minimize seepage of leachate into the subsurface (Kubare et al., 2010). There is no leachate collection system to store leachate generated from the landfill or methane collection system to trap methane gas emitted through an anaerobic waste decomposition in the landfill. Waste disposed at the site was not sorted and as such, hazardous waste could have been deposited at the landfill. During its operation, there was no compaction of

waste and covering of waste with soil at the end of each day as is required of landfill practice and waste was randomly disposed at the landfill. After the closure of the site, waste was compacted and covered with coal ash from a coal-fired power plant in the city (Teta and Hikwa, 2017). The site was also unrestricted or not fenced which means during its operation, scavengers or recyclers collected waste from the site.

Lumberstewart landfill used to be the main disposal site for the entire city before its complete closure in 1990. The landfill was in use for 15 years. Thickness of the coal ash used the landfill is approximately 30 cm and there is a probability that the ash could have been blown by wind to the surrounding area. The main sources of waste at the landfill were domestic and industrial facilities. Domestic waste in the form of unwanted household materials such as batteries, used light bulbs, old clothes, garbage, food wastes, ashes residues, plastics, papers and furniture were disposed at the site. Industries in Bulawayo produce automobiles, tyres, television sets, textiles, furniture and food. The city's population is approximately 653 337 (Zimstat, 2012) and on average produces approximately 325 tonnes of waste daily which is now disposed at Richmond municipal landfill. During its operation, about 280 tonnes of waste was dumped on a daily basis at the closed landfill.

3.2 Ethical Clearance

Permission to collect samples from the closed landfill site was obtained from the Bulawayo City council in Bulawayo Metropolitan Province (see Appendix 1). During sample collection, efforts were made to reduce the disturbance of the environment. Pits created during sample collection were backfilled and no endangered plant species were used in the study. In the analyses of the samples, laboratory safety precautions were employed and all waste chemicals and soils were disposed according to the laboratory practice. The study was issued an ethics certificate by UNISA reference number 2018 /CAES/009 (see Appendix 2).

3.3 Soil and Plant Sampling

3.3.1. Collection of soil samples

A systematic sampling approach was used to collect soil samples around the landfill. Soil samples were collected at various points in each of the cardinal directions of the landfill (that is north,

south, east and west). Three sampling sites were located along each of the north, south, east and west boundaries of the landfill, giving a total of 12 sampling points at the landfill. Each of the 12 sampling points was located 50m away from each other. Figure 3.4 below shows a sketch map of the landfill and the sampling points.

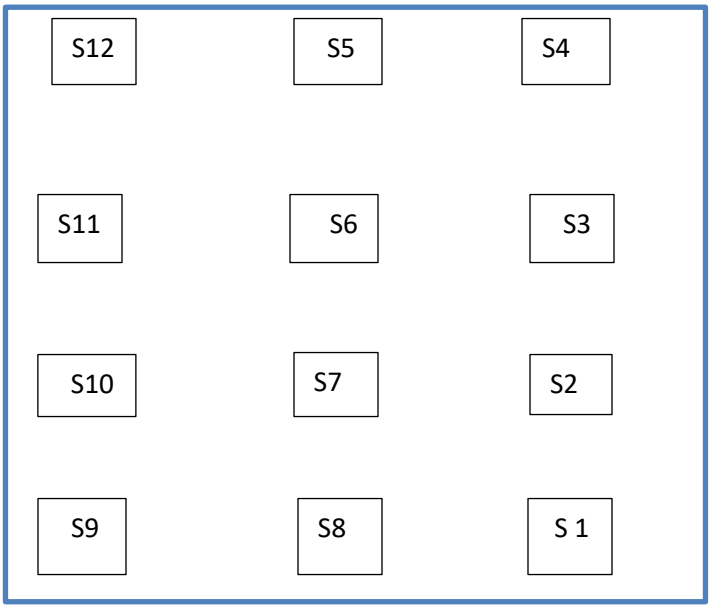


Figure 3.4: Sketch map of the landfill and the sampling points

At each of the 12 sites, soil samples were collected at a depth of 0-30 cm, 30-60 cm and 60-90 cm. The upper layers at the landfill were mainly made up of compacted sandy and coal ash whereas the bottom layers were composed of thin compacted clay soils. All soil samples were collected using a pick, spade and hand trowel. The trowel was washed with distilled water after collection of each sample before using it at the next sampling site to avoid cross contamination of samples. The procedure was repeated for control site which was located about 2km from the landfill site. The control site had the same geology and soils as the landfill site. In total 39 soil samples were collected for this research. Soil samples were transported to the laboratory in cardboard boxes. The samples were put in marked trays and were placed in racks in a hot-air cabinet at 35^{o}C for drying. Samples were then ground with a wooden pestle and porcelain mortar to disaggregate the particle. After grinding, the soil samples were passed through a 2mm sieve and placed in clean plastic bags, sealed and stored at room temperature for analysis.

3.3.2 Collection of plant samples

Dominant plant species with various uses in the community namely jimson weed (*Datura stramonium*) and pigweed (A*maranthus*) were collected from each site where soil samples were collected including the control site. At the control site same plants (jimson weed and pigweed) were collected A total of 26 plant samples were therefore collected for this study. The heavy metal concentrations of the plants were determined to have an idea of heavy metal transfer coefficient between the soil and plants in the study area. In the collection of plant samples, plants were uprooted and put in white plastic papers, marked and sealed before transportation to the laboratory. Prior to analysis, plants were washed with distilled water to remove soil particles from the roots and put in a well-ventilated oven for drying at 60℃ for 24 hours. After drying, the samples were grounded to pass through a 2mm mesh wire. The samples were stored in air-tight conditions until analysis of heavy metals (Estefan et al., 2013).

3.4 Soil Samples Analyses

Both physical and chemical properties of all the soil samples collected were determined. Physico-chemical properties analyzed in the soil samples included pH, electrical conductivity, texture, organic matter and cation exchange capacity. The concentrations of selected heavy metals including Cu, Cd, Cr, Zn, Fe and Ni in the soil and plant samples were also determined.

3.4.1 Soil texture determination

To determine soil texture, the hydrometer method was used. From each dried and sieved soil sample, 50 g was weighed into a beaker and 100 ml of 0.05% sodium hexametaphosphate solution added to disperse the soil particles. After this, 250 ml of distilled water was then added to the soil: sodium hexametaphosphate suspension and mixed for five minutes with an electric mixer. The mixture was poured into a 1000 ml (1L) sedimentation cylinder and distilled water added to reach the 1L mark (Estefan et al., 2013). Method calibration with a blank was done by adding 100 ml of solution to 900 ml distilled water in another cylinder without soil and the mixture homogenized with a plunger. The temperature of the soil suspension and the blank were noted. The mixtures were homogenized, placed on a steady surface, a hydrometer immediately inserted, and the

hydrometer readings recorded after 40 seconds. This was repeated three times (Estefan et al., 2013). This reading represented the density of the suspension of silt plus clay in the sample as it is believed that sand particles would have settled at the cylinder bottom during this period. The suspension was again homogenized, placed on a steady surface and allowed to stand for six hours. After 6hrs, another hydrometer reading was recorded and this reading represented the density of a suspension of clay as sand and silt particles would have settled to the bottom of the cylinder during this time. The weight percentage of sand, silt and clay in each soil sample was calculated as indicated in equations 1, 2 and 3 (Estefan et al., 2013).

$$\%\, Clay = \frac{Corrected\ hydrometer\ readint\ at\ 6\ hrs}{weight\ of\ sample} \times 100 \qquad \text{Equation (1)}$$

$$\%\, Silt = \frac{Corrected\ hydrometer\ readint\ at\ 40\ seconds}{weight\ of\ sample} \times 100 \qquad \text{Equation (2)}$$

$$\%\, Sand = 100\% - \%silt - \%clay \qquad \text{Equation (3)}$$

3.4.2 Determination of soil pH

To determine soil pH, 10g of each air-dried and sieved soil sample were transferred into separate conical flask. Distilled water (25 ml) was poured into the soil sample to obtain a 1:2.5 soil to water weight/volume ratio (Kabala et al., 2016). The mixture was placed on a multi-purpose rotator (Barnstead Lab Line) and stirred for one hour. A microprocessor pH meter (HANNA Instruments) was calibrated with two buffer solutions, at pH 4.0 and 7.0. After stirring the samples, the pH probe of the pH meter was immersed in the upper part of the soil: water suspension and the pH readings taken. Readings were recorded only when the readings on the pH meter had stabilized. According to Ugwoha and Emete (2015), the reading of pH is stable when it does not change more than 0.1 per 30 seconds or 0.02 units per 5 seconds. This was repeated three times for each sample and the mean of the three readings recorded as the final pH of each sample.

3.4.3 Determination of soil electrical conductivity

To determine the electrical conductivity of the soil samples, 50 ml of distilled water was added to a beaker with 10 g air-dried and sieved soil sample to obtain 1:5 soil to water ratio (Estefan et al.,

2013). The mixture was shaken vigorously for about 25 seconds and allowed to stand for a while. A calibrated EC meter was inserted in the suspension of soil and water (Susu and Salami, 2011). The reading was taken while particles were suspended in solution. Two readings were taken from each sample and the average was calculated and recorded as the electrical conductivity of the soil sample.

3.4.4 Determination of soil organic matter content

The Loss-On-Ignition (LOI) method was used to determine soil organic matter. Porcelain crucibles were heated for one hour at 375^0C in the furnace and cooled to about 150^0C. Soil samples sieved with a 2 mm sieve were oven dried at 105^0C for 24 hours to remove moisture (Luke et al., 2009). Each 5 g dried soil sample was then weighed into the pre-weighed porcelain crucibles (pre-ignition weight) and then placed in the muffle furnace at 550^0C. The soil samples were heated in the furnace for 16 hrs after which they were removed and weighed again (post-ignition weight). The porcelain crucible weight was subtracted from the post-ignition weight. Organic matter content was calculated as indicated in equation 4 (Luke et al., 2009).

$$Organic\ matter\ \% = \frac{Pre{-}ignition\ weight\ (g){-}Post{-}ignition\ weight\ (g)}{Pre{-}ignition\ weight\ of\ soil\ sample} \times 100 \qquad \text{Equation (4)}$$

3.4.5 Determination of soil cation exchange capacity

The Ammonium acetate (NH_4AOc) method was used to determine cation exchange capacity of the soil samples. Each 10 g soil sample which had been dried and sieved was weighed into a 500 mL Erlenmeyer flask with a stopper and 250 mL of ammonium acetate added (Mohamedharoonbasha and Chandramohan, 2012). The mixture was then shaken thoroughly for five minutes and allowed to stand overnight. The following day, the mixture was shaken for 15 minutes on a rotary shaker. An erlenmyer flask with a side arm was hooked to a vacuum pump and a funnel placed on the mouth of the flask. The funnel was lined with Whatman No. 42 Filter paper to avoid soil particles entering the flask. The soil-ammonium acetate suspension was poured into the funnel and the filtrate sucked into the vacuum flask. The soil was rinsed with ammonium acetate to wash off exchangeable ions displaced by the NH_4^+. The samples were washed out with 200 ml of 99 % isopropyl alcohol and the soil was allowed to drain. Sodium chloride was used to displace the adsorbed NH_4 from the soil by leaching the soil with 10 % acidified NaCI until 225 ml of NaCl had passed through the soil sample. The leachate was transferred to an 800 ml Kjeldah flask, 25

ml of NaOH and 60 ml of the solution was distilled into a 50 ml of 2% H_3BO_3. 10 drops of bromocresoil green-methyl red mixed indicator were added and the boric acid solution was titrated with standard 0.1 H_2SO_4. Blanks were run on the reagents and correction of titration for the blanks was done. The CEC of the soil was calculated according to Equation (5) (Gillman, 1979).

$$CEC = \frac{(vol\ of\ titre\ used\ for\ sample - vol\ of\ titre\ used\ for\ blank) \times N\ \times 100 \times mcf}{weight\ of\ sample\ used}$$

Where	CEC	=	cation exchange capacity (meq /100g
	N	=	normality of acid used for titration
	100	=	conversion factor to 100g basis
	mcf	=	moisture correction factor for each sample

3.4.6 Determination of soil heavy metal concentrations

To determine heavy metal concentrations in soil, soil samples were digested using the EPA Method 3050B: Acid Digestion of Sediments, sledges and soils. According to this method, 1 g of each soil sample was weighed into a digestion vessel into which 20 ml aqua regia (25% nitric acid: 75% hydrochloric acid v/v) was added and the resulting slurry mixed (Cortez and Ching, 2014). The slurry was covered with a vapor recovery device and heated to 95^0C to reflux for 10 minutes without boiling. After cooling of the sample, 2 ml of distilled water and 3 ml of 30% hydrogen peroxide (H_2O_2) were added until the effervescence was minimal or the sample's appearance was unchanged. The vessel was covered with the vapor recovery device and returned to the heat source again for warming to start the H_2O_2 reaction. The Acid peroxide digestate was heated at 95^0C until the volume reached 5 ml. After cooling, distilled deionized water was added into the digestion tube to dilute the digestate to 100 ml. Particles in the digestate were removed by filtration using Whatman No.41 filter paper (Ogundiran and Afolabi, 2008). The filtrate was analyzed using Flame Atomic Absorption Spectrometry (AAS) (John et al., 2009; Ogundiran and Afolabi, 2008, Cortez and Ching, 2014). Atomic Absorption Spectrometry conditions used for metal analyses are presented in Table 3.1 below:

Table 3.1: Atomic Absorption Spectrometry conditions used for heavy metals analyses

Metal	**Wavelength**	**Slit diameter**	**Lamp Current**	**Flame type**

	(nm)	(nm)	(Am)	
Fe	248.3	0.2	6	Air-acetylene
Zn	213.9	0.7	5	Air-acetylene
Cu	324.7	0.4	5	Air-acetylene
Cr	357.9	0.2	7	Air-acetylene
Ni	232.0	0.7	7	Air-acetylene
Cd	228.8	0.7	4	Air-acetylene

High purity metal standards approved and endorsed by National Institute of Standards and Technology (Fluka* Analytical Sigma-Aldrich, Germany) were used for the calibration and checks after analyzing every five samples (Teta and Hikwa, 2017). Concentration of heavy metals in the soils samples were calculated as shown in Equation (6).

$$Concentration\ (mg/kg) = \frac{Concentration\ (mg/L) \times V}{W}$$

Where V=Final Volume of solution and

W= initial weight of sample measured

3.5 Plant Samples Analysis

The Nitric acid-peroxide method was used for digestion of plant samples (Estefan et al., 2013). To 5 g of each ground dried plant samples weighed into a conical flask, 5 ml of 65% nitric acid and 5 ml of 30% hydrogen peroxide were added and the mixture placed on a hot plate. The mixture was heated at a temperature of 140 °C for about 45 minutes until the plant material dissolved and the contents reduced to 5 ml (Belay, 2014). The contents were filtered through an acid washed filter paper, transferred to a 50 ml volumetric flask, and made up to volume by adding de-ionized water. A Flame Atomic Absorption Spectrometer (AAS) with the same set up parameters as indicated in Table 3.1 was used to determine heavy metals in plants digests**.**

3.6 Quality Assurance/ Quality Control Measures

To ensure quality during the study, standard plant and soil reference materials endorsed by NIST for selected metals were used. The Standard Reference Materials (SRMs) used were Cd (3108), Cr (3112a), Cu (3114), Fe (3126a), Ni (3136) and Zn (3168). Results of the concentrations of the different heavy metals in the reference sample obtained from the equipment were compared with the known values to determine percent heavy metal recovery by the equipment. All samples were analyzed in duplicates and the means used as final values to ensure accuracy. All re-agents used for metal analysis were of analytical grade from MERCK Germany. Instrument calibration using certified standard was done after every five samples. Glassware was cleaned by soaking and rinsing in acidified distilled deionized water (1% nitric acid) before use.

3.7 Data Analysis

Descriptive statistics including mean and standard deviation of the concentration of heavy metals in the samples analyzed was determined using SPSS program. ANOVA with Tukeys test of significance was used to determine the differences in heavy metal concentrations between sites, differences in soil properties between site and between depth and the difference in heavy metal concentrations in plants from one site to the other. The differences in heavy metal concentrations in the two plant species studied were also compared using student-t-test. All statistical analyses were carried out at a 95% confidence interval. To determine the extent of heavy metal contamination in soil at the landfill, heavy metal contamination factor (CF) for each of the metals analysed was calculated according to Equation (7).

$$CF = \frac{\text{Cm Sample}}{\text{Cm Background}} \qquad \text{Equation (7)}$$

Where Cm Sample = concentration of a metal in a landfill and

Cm Background = concentration of that metal in background samples (which in this case was the control sample).

According to Sutherland, et al. (2000), values of heavy metal CF < 1 refer to low contamination; $1 \leq CF < 3$ implies moderate contamination; $3 \leq CF \leq 6$, considerable contamination, and CF > 6 indicates very high contamination. The Pollution Load Index (PLI) which indicates the extent of

heavy metals contamination in soils was also calculated to determine which of the sites was the most contaminated when all heavy metals analyzed are taken into consideration (Al-Juboury, 2009). PLI values <1 indicate no pollution whereas values >1 indicate pollution. The PLI for each site was calculated according Tomlinson et al. (1980) as shown in Equation (8).

$$\text{PLI} = \sqrt{(CF1xCF2xCF3 \ldots\ldots xCFn)} \qquad \text{Equation (8)}$$

Where, CF is contamination factor and

N is the number of elements.

To determine the uptake of heavy metals by plants, from soils around the landfill, heavy metal transfer factor was calculated for each metal as indicated in Equation (9)

$$T.F = \frac{Cplants}{Csoils} \qquad \text{Equation (9)}$$

Where T.F is the Transfer Factor,

C_{plants} is heavy metal concentration in plants and

C_{soils} is heavy metal concentration in soil.

T-test assuming equal variance was employed in Microsoft excel to determine whether differences between sites and samples were significant or not. Pearson correlation between heavy metal content in plant and soil was calculated using excel to determine whether there were any relationships between the concentrations of heavy metals and the properties of soils around the landfill.

CHAPTER 4

RESULTS AND DISCUSSION

4.1 Introduction

This chapter presents results of the laboratory analyses which were carried out on the plant and soil samples from the Lumberstewart landfill.

4.2 Properties of Soils at the Landfill

4.2.1 Soil Texture

Results from particle size analysis indicate that soils from the landfill had a high content of sand ranging between 61.6% -76.5%, followed by clay particles with a range of 11.5% - 23.2% and then silt particles with a range of 9.30%-17.1% (Table 4.1). Soils from sites 8 and 9 at the Lumberstewart landfill had the highest amount mean sand content whereas sites 1, 4 and 12 had the lowest. Clay content was higher in soils from sites 1, 3, 4 and 11 at the landfill compared to the other sites ($P < 0.05$). The control site had a higher percentage of sand (81.4%) and silt (10.2%) compared to sites around the landfill. Clay content in soils at control site was less than that of the landfill averaging 12. 5%. Sand content in the control site was significantly higher than that at sites 1, 2, 3, 4, 5, 10, 11 and 12 ($p < 0.05$), whereas the silt and clay contents were lower than those in sites 1, 2, 3 4 and sites 1, 3, 4, 10 and 11 respectively ($P<0.05$). Sand content in the soils decreased from a depth of 0-30 cm to a depth of 60-90 cm, whereas clay content increased with depth (figures 4.1 – 4.3). This might indicate an increase in base content of the underlying granite rock which according to Vennum and Nejedly (1990) and Price (1995) weathers chemically to produce clay

minerals. The observed texture (Figure 4.4) may be due to natural weathering processes going on around the landfill but the clay content at depths of between 0 – 30 cm might also have been affected by the coal ash which was used to cap the landfill, coal ash is fine textured and could have been blown to the surrounding soils.

Table 4.1: Mean concentrations of sand, silt and clay around the landfill

Site	Sand	Silt	clay
Site 1	64.3±1.69	15.4±0.55	20.3±1.51
Site 2	68.6±1.44	14.5±0.59	16.9±1.04
Site 3	64.6±1.05	16.3±0.70	19.1±0.85
Site 4	64.3±0.78	14.6±1.12	21.1±0.39
Site 5	72.9±0.88	11.8±0.27	15.3±0.67
Site 6	75.5±0.69	10.0±0.29	14.4±0.55
Site 7	70.4±0.75	11.3±1.60	18.3±1.04
Site 8	75.7±0.64	10.0±0.35	14.3±0.99
Site 9	75.9±0.48	10.7±0.10	13.4±0.38
Site10	72.4±1.31	10.2±0.40	17.4±1.62
Site11	70.1±1.17	10.5±0.54	19.4±1.62
Site 12	64.3±2.13	12.2±2.11	12.5±0.50
Control site	79.9±0.87	8.2±1.04	11.9±0.32

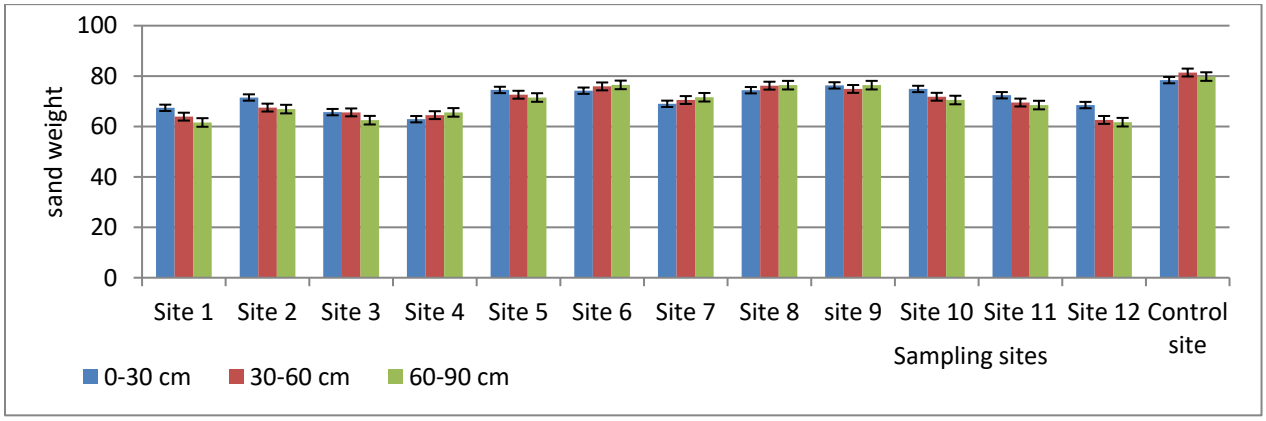


Figure 4.1 Sand content in soils from different depths at the landfill

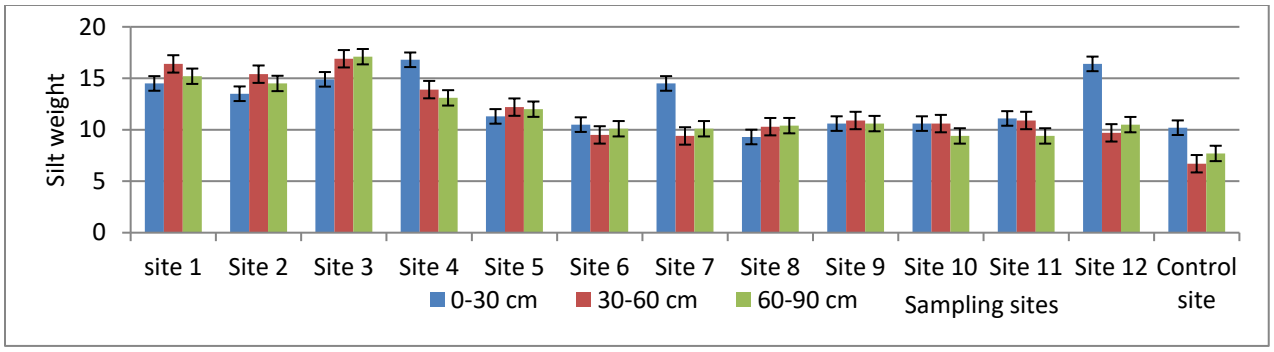


Figure 4.2 Silt content in soils from different depths at the landfill

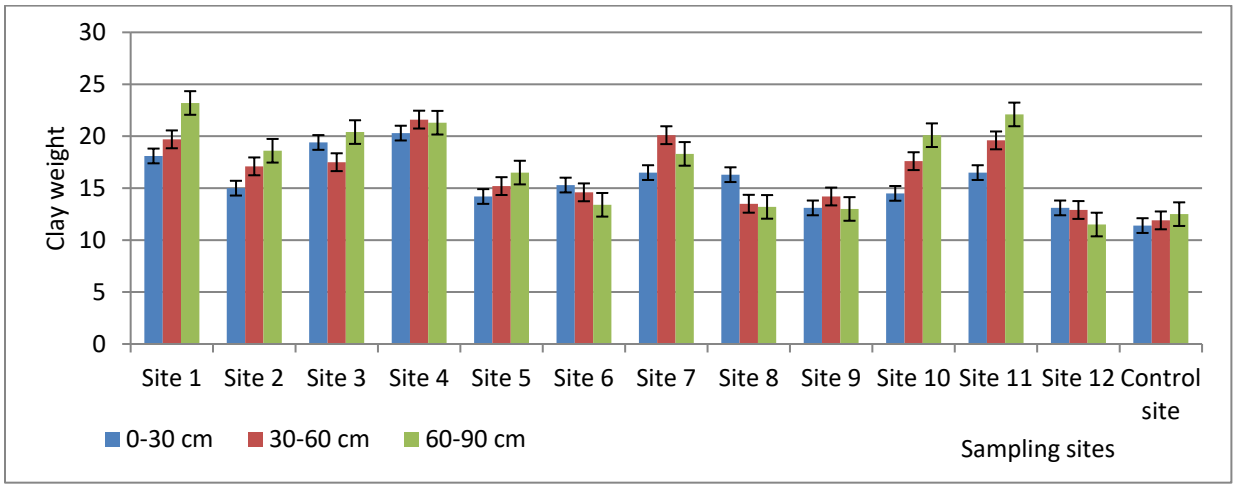


Fig 4.3 Clay content in soils from different depths at the landfill

According to Evanko and Dzombak (1997), high amount of sand in soils increases soil porosity due to large pore sizes present in coarse textured soils whereas high content of clay in soils reduces water movement since in clayey soils particles are closely packed hindering the movement of water. The sandy loam and sandy clay loam texture characteristic of soils at the sampled depths at the landfill site (Figure 4.4) could facilitate leachate movement from the landfill into surrounding environments. Madzhieva et al. (2014) reported that the sorption of heavy metals increases with decreasing particle size. Coarse textured soils like sandy soils have low sorption capacity while fine textured soils like clay have high soil sorption capacity due to enrichment of soil components with high surface area and negative charge density. This could also reduce the spread of landfill-derived heavy metals in the soils around the landfill hence reducing the availability of these metals and their eventual uptake by plants growing in the area.

4.2.2 Soil pH

The pH values of 0-30 cm depth ranged between 5.50 (site 11) to 6.55 (site 6) which revealed that the surface soils at the landfill were slightly acidic. At a depth of 0-30 cm soil pH values were lower than the pH values of soils at a depth of 30-60 cm ($p = 0.02$) with pH values of soils at 30-60 cm ranging from 6.02 in soils from site 3 to 7.65 in soils from site 5, which is nearly an alkaline pH (Figure 4.5). Soil pH at a depth of 60-90 cm ranged from 5.01 - 6.60 which is lower than the pH of the soils from 0-30 cm depth and soils at a depth of 30-60 cm ($p = 0.01$) (Figure 4.5). The differences observed in soil pH between the different soil depths were significant ($p<0.05$).

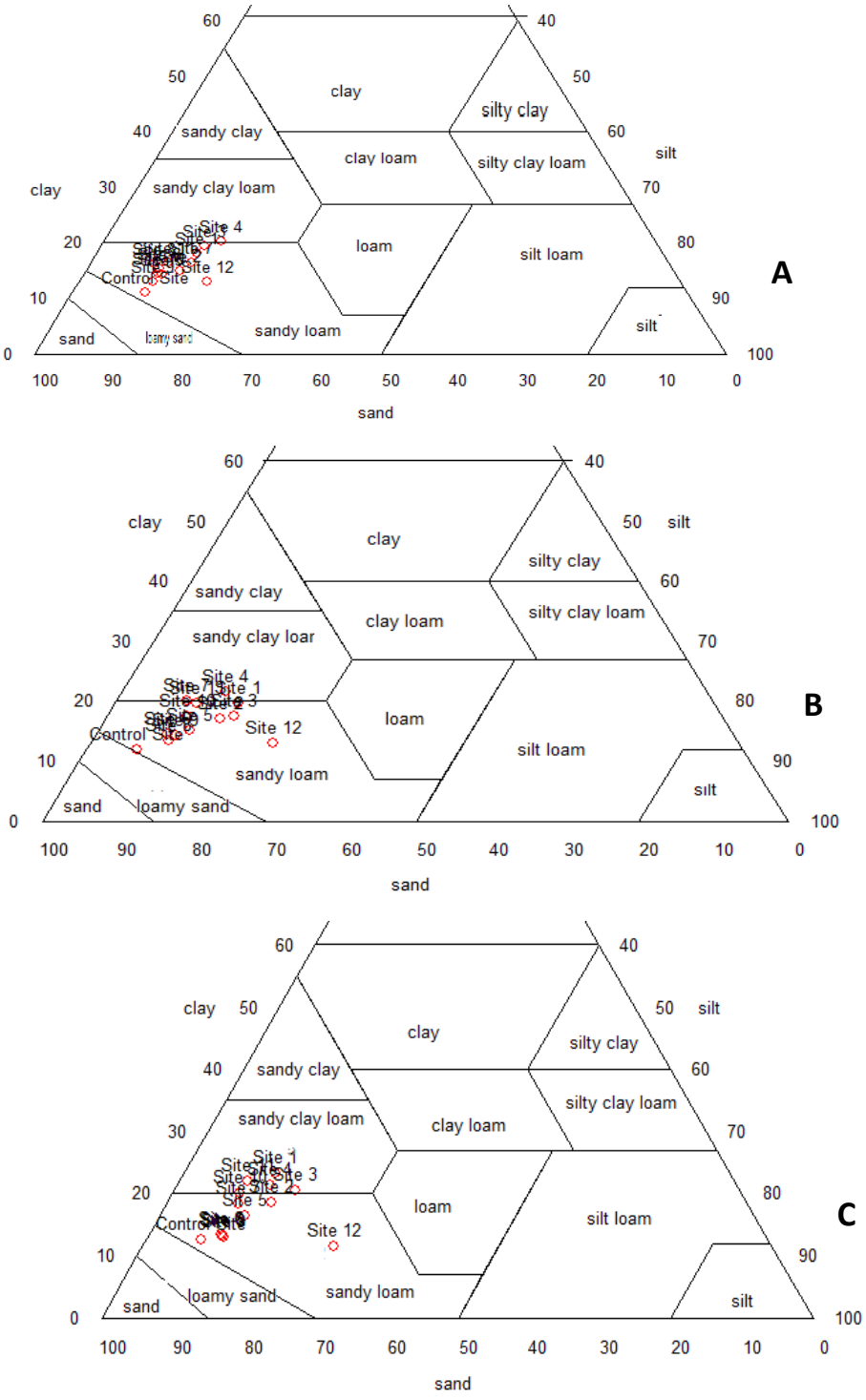


Figure 4.4: Textural triangles showing the texture of the soils at a depth of 0-30cm (A), depth of 30-60 cm (B) and depth of 60-90 cm (C) at the landfill

Comparatively, soil pH at the control site was alkaline as compared to the landfill (Fig 4.5). Though the mean pH of soils around the landfill were lower than what was obtained at the control site, the differences were insignificant (P>0.05). Results of soil pH obtained in this study revealed that the soils are acidic and this is similar to studies by Salami et al. (2013) but contrary to what was reported in a landfill by Hunachew and Sandip (2011). Old landfill sites are characterized by high soil pH values as compared to young landfills since pH increases with time because of a reduction in concentration of the partially ionized free volatile fatty acids produced during waste decomposition (Madzhieva et al., 2014). The level of alkalinity in soils from the Lumberstewart landfill is however lower compared to typical pH values of soils around old landfills. For most of the sites, the pH increased at a depth of 30 – 60 cm after which pH values again dropped (Figure 4.5). According to Zhang (2002), organic waste materials at the landfill decompose and produce hydrogen ions responsible for formation of acidic conditions. Basic cations such as potassium (K^{+}), magnesium (Mg^{2+}) and sodium (Na^{+}) are leached by moving water out of the waste and replaced by acid forming cations such as aluminum (Al^{3+}) which increase the acidity of the soils.

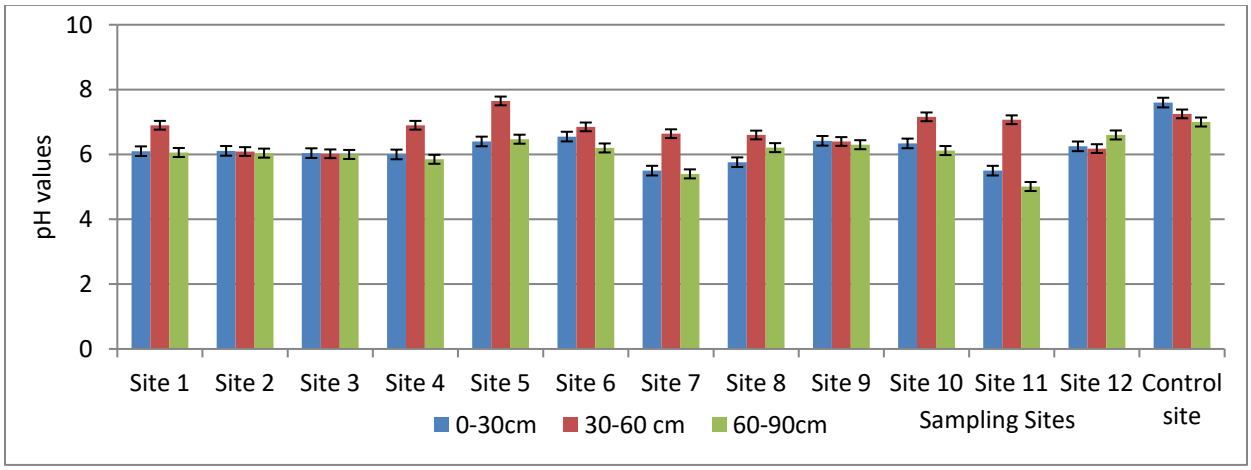


Figure 4.5: Soil pH at different depths at the landfill

The decrease in pH between 60 and 90 cm depth at Lumberstewart landfill can be attributed to high rate of evapotranspiration in the area which causes salts such as sulphates, chlorides, nitrates and bicarbonates to precipitate on the subsurface horizons of soils (Alloway and Ayres, 1997). The 60 – 90 cm depth could be the zone where ions are moving toward the soil surface due to capillary action whereas the 0 – 60 cm could represent the zone where leachate is percolating the soil.

The pH of the soils at the landfill is likely to increase the availability of nutrients to plants in the vicinity of the landfill especially around sites 7 and 11 which have the lowest mean pH values around the landfill (Table 4.2). This could increase the amount of heavy metals taken up by the plants affecting their growth. According to de Souza et al. (2011), acid soils could reduce the growth of pigweed and jimson weed as a result of the reduction of important plant nutrients. This is evident by unhealthy jimson weed leaves at the landfill site.

Table 4.2: Mean concentrations of soil pH around the landfill

Site	Mean ± Standard error
Site1	6.35±0.27
Site 2	6.08 ±0.02
Site 3	6.02 ±0.01
Site 4	6.25 ±0.33
Site 5	6.84 ±0.41
Site 6	6.53 ±0.19
Site 7	5.85 ±0.40
Site 8	6.19 ±0.24
Site 9	6.37 ±0.04
Site 10	6.54 ±0.32
Site 11	5.86 ±0.62
Site 12	6.34 ±0.13
Control site	7.28 ±0.17

4.2.3 Soil Electrical Conductivity

Electrical conductivity of the soil samples collected at the landfill ranged between 0.43 to 1.67 dS/m at a depth of 0-30 cm, 0.50 to 1.5 dS/m at a depth of 30-60 cm and 0.50 and 1.60 dS/m at a depth of 60-90 cm (Figure 4.6). The soil electrical conductivity was higher at a depth of 60-90 cm as compared to a depth of 0-30 cm and soils from a depth of 30-60 cm. However, the differences between soil depths were not significant ($p>0.05$). At a depth of 30-60 cm, values of EC increased slightly with the highest EC value recorded at site 8 (1.56 dS/m) whereas the lowest value was

recorded at site 7 (0.50 dS/m). Soil samples from site 6 had the highest EC values (1.60 dS/m), whereas those from site 11 had the least EC values (0.56 dS/m) at a depth of 60-90 cm. Results indicate that electrical conductivity increased with depth and this could be due to accumulation of salts and ions washed down from the decomposing waste by percolating water (Praveena and Prasado, 2016).

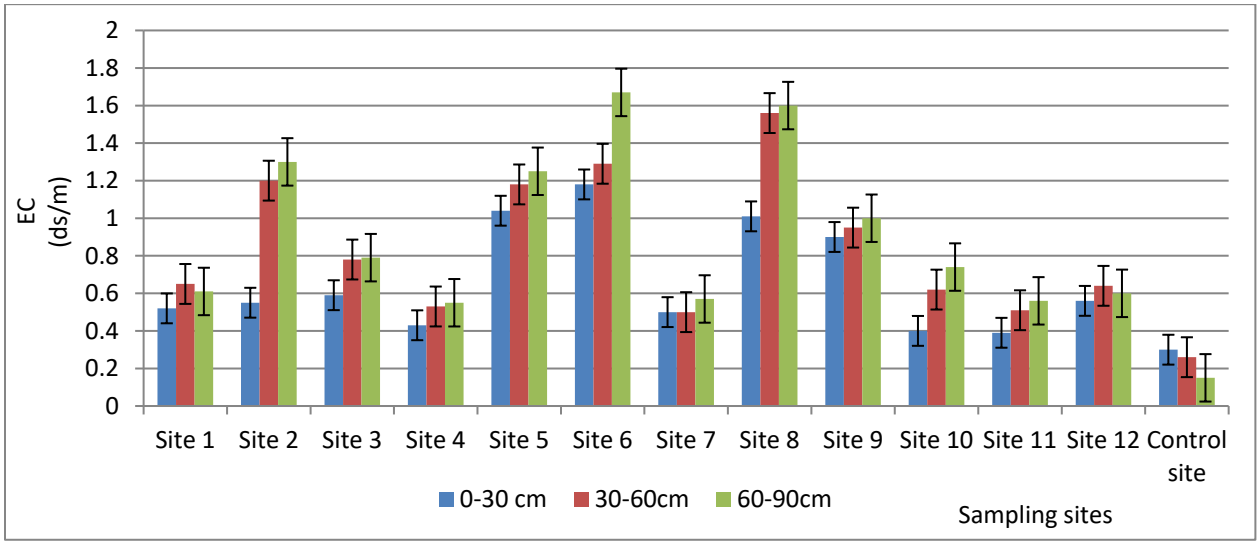


Figure 4.6: Soil electrical conductivity (dS/m) at different depths at the landfill

Results of EC obtained from soils at sites 2, 5, 6, 8 and 9 at the closed Lumberstewart landfill were significantly higher ($P < 0.05$) than those from soils at the control site (Figure 4.6). Soils from sites 2, 6 and 8 had the highest mean of EC values whereas sites 4, 7 and 11 had the lowest (Table 4.3). Statistical test for difference in EC indicate significant differences in EC values between sites ($P < 0.05$) but no differences with depths ($P > 0.05$). Electrical conductivity indicates the level of salinity of soils with high EC values indicating high levels of salinity. These salts increase free moving electrons which contribute to the soils' electrical conductivity, especially when the soil is wet (Goswami and Sarma, 2008). The EC values indicate that all sites had non-saline conditions as the EC of soils at all sites were < 2 dS/m (Shrivastava and Kumar, 2018). According to Goswami and Sarma (2008), electrical conductivity values less than 0.50 dS/m are safe and do not have negative effects on the growth of plants. This study revealed that the salinity levels of the soil samples from 0-30 cm, 30-60 cm and 60-90 cm depths were low hence suitable for agricultural

activities (Smith and Doran, 1996). High salinity levels are not likely to be a challenge to plants growing at the closed Lumberstewart landfill.

Table 4.3: Mean concentrations of EC around the landfill

Site	Mean ± Standard error
Site 1	0.59±0.04
Site 2	1.02±0.24
Site 3	0.72±0.07
Site 4	0.50±0.04
Site 5	1.16±0.06
Site 6	1.38±0.15
Site 7	0.52±0.02
Site 8	1.39±0.19
Site 9	0.95±0.03
Site 10	0.59±0.10
Site 11	0.49±0.05
Site 12	0.60±0.23
Control site	0.24±0.04

4.2.4 Soil Organic Matter Content

Site 9 had the highest amount of organic matter (2.50%) while site 6 had the least amount (0.8%) at all depths compared to other sites (Figure 4.7). High organic matter content was obtained in soils from 0-30 cm and 60-90 cm depths with OM values ranging between 1.00 - 2.50% and 0.98 - 2.25% respectively At a depth of 60-90 cm, organic matter was higher (0.98- 2.25%) than at a depth of 30-60 cm (0.80-2.00%) at the landfill and this can be attributed to activities such as leaching. Soils from the control site had lower amounts of organic matter recording 1.50% at 0-30 cm, 1.20% at 30-60 cm and 1.00% at 60-90 cm. Soil organic matter in the control site was significantly lower than that at the landfill at sites 4, 9, 10, 11 and 12 ($p< 0.05$). Sites 4, 9, 11 and 12 at Lumberstewart landfill had the highest mean whereas site 6 had the lowest (Table 4.5). A comparison of the OM content in soils at different sites using Student t-test indicated significance differences in soils organic matter content between soils at a depth of 0-30 cm and those at a depth

of 30-60 cm ($P < 0.05$). On the other hand, there were no significant differences between organic matter content in soils at depths of 30-60 cm and 60-90 cm ($p > 0.05$).

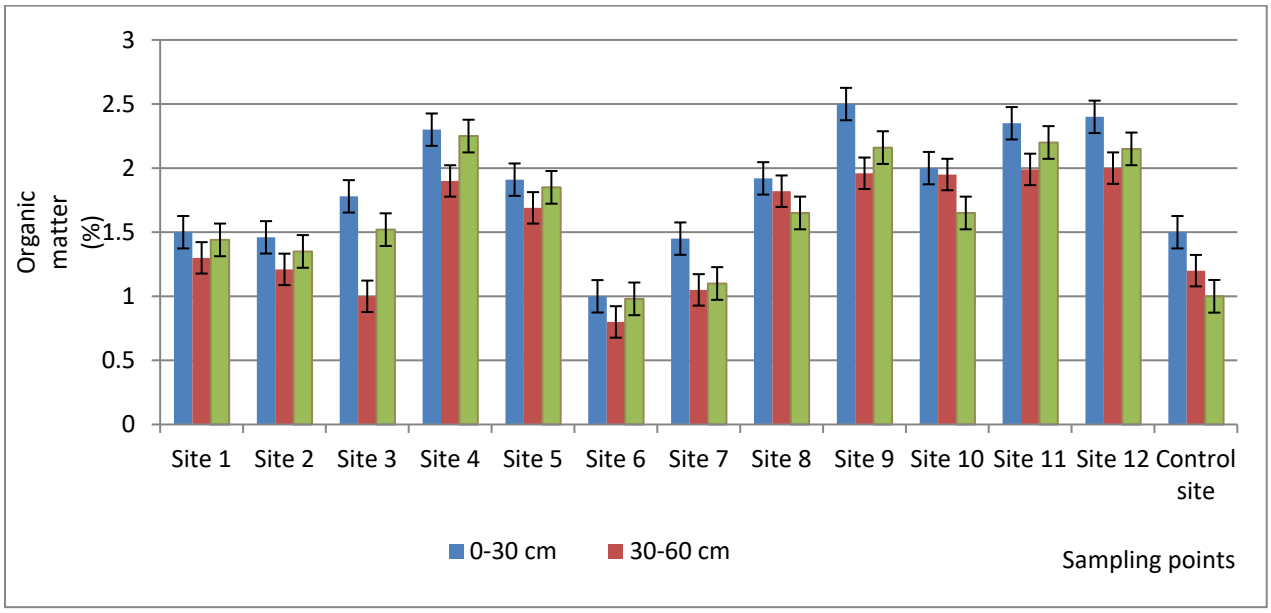


Figure 4.7: Organic matter content of soils at different depths at the closed landfill

Table 4.4: Mean concentrations of organic matter around the landfill

Site	Mean ± Standard error
Site 1	1.41±0.06
Site 2	1.34±0.72
Site 3	1.43±0.23
Site 4	2.15±0.13
Site 5	1.82±0.07
Site 6	0.93±0.06
Site 7	1.20±0.13
Site 8	1.80±0.08
Site 9	2.21±0.16
Site 10	1.87±0.11
Site 11	2.18±0.10
Site 12	2.18±0.12

Control site	1.23±0.15

At a depth of 0-30 cm, high organic matter could have accumulated as a result of waste containing materials and residues from dead animals, plants and plant roots. According to Tripathi and Misra (2012), high organic matter at the surface is mainly because of the presence of organic residues from agricultural activities and some recreational activities which add more organic matter after decomposition of litter. In this study very weak negative correlation was recorded between organic matter and sand at 0-30 cm (r =0.01), at 30-60 cm depth (r = 0.02) and at 60-90 cm (r = 0.19). Results also indicated that there were very weak negative correlation between organic matter and clay soils at all depths (r = -0.15, r = -0.13, r = -0.01). Organic matter and silt in the soils also recorded very weak positive correlation at a depth of 0-30 cm r = 0.08) whereas the other depths, 30-60 cm and 60-90 cm recorded very weak negative correlations (r = -0.30, r = -0.16). Lack of correlation between OM content in the soils and the weight percent of the different soil particles highlights that the increase in OM is of anthropogenic and not natural origins.

4.2.5 Soil Cation Exchange Capacity

Cation exchange capacity of the soils at a depth of 0-30 cm of the landfill was highest at site 1 (15.7 cmol kg^{-1}) followed by those from site 3 (15.6 cmol kg^{-1}) while soils from site 11 had the lowest CEC value (4.30 cmol kg^{-1}) at 60-90 cm. At a depth of 0-30 cm, soil CEC ranged between 5.30 and 15.7 cmol kg^{-1} whereas at 30-60 cm depth, the CEC values of the soils were slightly lower, ranging between 5.00 and 14.4 cmol kg^{-1}(Figure 4.8). Soils at a depth of 60-90 cm had CEC values which were almost similar to those at a depth of 30-60 cm with CEC values ranging from 4.45 - 14.0 cmol kg^{-1} (Figure 4.8). Soil CEC in the control site was lower than that at sites 1, 2, 3, 4, 5, 6, 7, 8, 9 and 10. Despite the soils from the control site having lower CEC values than soils from the landfill, results obtained through student t-test which was used to compare the CEC values of the different samples indicate that there were no differences in CEC values at different depths around the landfill (P > 0.05). Soils from sites 1 and 3 had the highest amount mean CEC whereas sites 11 and 12 had the lowest (Table 4.5). Generally, the CEC of the soils are within the range of values reported for sandy soils which dominate the soils around the landfill. According to Aydinalp and Marinova (2003), soil CEC increases with OM and clay content in soils due to an increase in the density of negative charges in such soils. Sand particles are relatively chemically inert with

low surface charge density and a low capacity to retain cations. The low soil CEC observed in the soils at the landfill could also be attributed to the low organic matter content in the soils. Low CEC of the soils at the Lumberstewart Landfill could imply high mobility of heavy metals in the soils and their eventual uptake by plants growing in the vicinity of the landfill (Aydinalp and Marinova, 2003). It could also imply that nutrients are continually leached from the soil, reducing its fertility and capacity to produce acceptable yields.

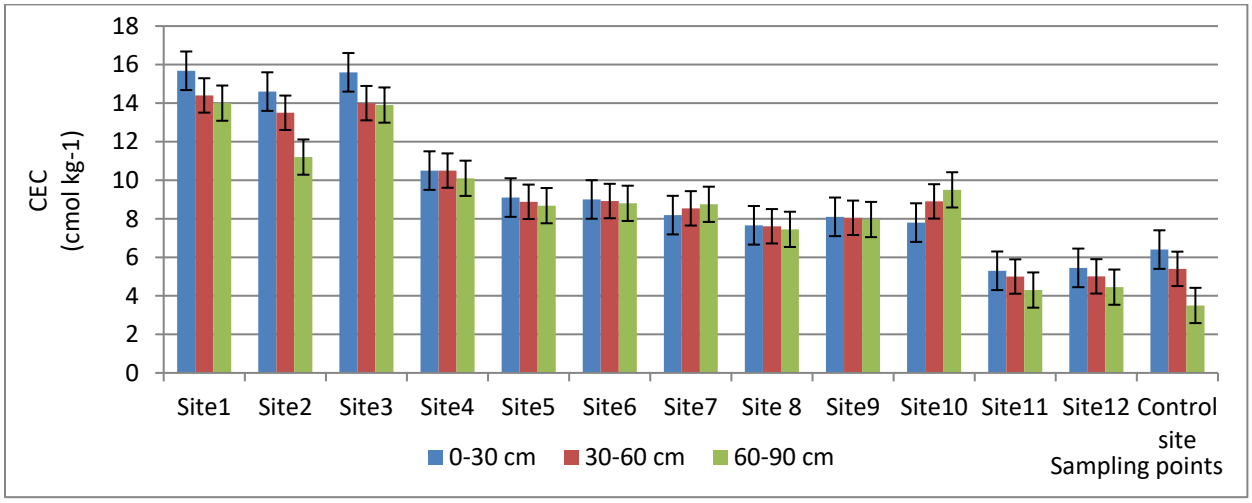


Figure 4.8: Cation exchange of soils at different depths at the landfill

Table 4.5: Mean concentrations of CEC around the landfill

Site	Mean ± Standard error
Site 1	14.7±0.48
Site 2	13.1±1.00
Site 3	14.5±0.55
Site 4	10.4±0.13
Site 5	8.89±0.12
Site 6	8.91±0.06
Site 7	8.49±0.16
Site 8	7.57±0.06
Site 9	8.04±0.04
Site 10	8.73±0.50

Site 11	4.87±0.30
Site 12	4.97±0.29
Control site	5.10±0.85

4.3 Concentration of Heavy Metals around the Landfill Site

4.3.1 Concentration of Heavy Metals in Soils

The concentrations of the heavy metals studied in soils around the landfill varied with Fe having the highest concentration and Cd the lowest (Table 4.6). Cd concentration in soils was highest on site 1 averaging 1.00 mg/kg whilst the lowest concentration was observed in soils from sites 5, 8, 9, 10, 11 and 12 averaging 0.01mg/kg. Only few sites had soils with >0.01mg/kg for values of concentrations of Cd at all depths as shown at Fig 4.9. About half of the sampling sites had soil Cd concentration which was almost similar to the control site whereas the other half differed. The mean Cd of soils at the landfill was the same with what was obtained at the control site except for sites 1, 2, 3, 4, 6 and 7 which were significantly higher than the control site ($P<0.05$). (Table 4.6).

Table 4.6: Mean concentrations of heavy metals in soils around the landfill

Site	Cd	Cr	Cu	Fe	Ni	Zn
Site 1	0.40±0.23	29.6±8.38	127±40.6	44716±12189	28.0±1.97	288±78.6
Site 2	0.40±0.40	25.6±7.19	81.0±12.6	41294±12744	35.0±6.25	371±86.3
Site 3	0.37±0.36	24.0±6.70	78.2±13.8	42309±11612	34.3±6.34	342±84.1
Site 4	0.27±0.17	23.8±6.75	81.8±14.4	39392±8857	31.8±6.78	360±85.2
Site 5	0.01±0.00	40.6±2.40	93.6±15.4	32622±4428	25.6±3.37	267±134
Site 6	0.37±0.09	41.9±1.79	130±39.7	45691±12202	28.9±1.64	304±72.5
Site 7	0.51±0.26	41.5±3.53	129±40.1	44694±12206	28.5±1.80	297±74.8
Site 8	0.01±0.00	39.6±2.39	86.4±14.8	33645±7855	22.0±2.57	253±127
Site 9	0.01±0.00	37.2±1.16	79.0±13.3	39796±12784	30.9±6.68	349±84.3
Site 10	0.01±0.00	35.5±1.55	63.9±7.27	34849±10420	29.6±6.48	333±82.8
Site 11	0.01±0.00	35.3±1.69	57.7±8.08	34924±9924	29.4±6.40	228±81.1
Site 12	0.01±0.00	34.9±1.52	69.9±5.92	33238±7870	21.1±2.88	273±59.4

Control Site	0.01±0.00	11.7±0.33	22.4±1.81	12080±439	19.7±0.33	329±8.14
NEMA	7.5	6.5	16	NA	91	240
WHO	0.01	0.05	1.50	20.0	6.5	15.0

The concentrations of Cd observed in the soils at the Lumberstewart landfill are below the permissible limits in soils (7.5mg/kg) in South Africa (NEMA 2008; Mtunzi et al., 2015; Durowoju et al., 2016; SABS, 1999; DNHPD, 1991).

Cd values of 0-30 cm depth ranged between 0.01 mg/kg (sites 2, 3, 5, 8, 9, 10, 11, 12) and 0.50 mg/kg (site 6). At a depth of 0-30 cm, Cd values were higher than at a depth of 30-60 cm with Cd values ranging from 0.01 from soils at sites (1, 4, 5, 8, 9, 10, 11 and 12) to 1.20 mg/kg from site 2. Cd at a depth of 60-90 cm ranged from 0.01-0.80 mg/kg which is lower than in soils from 30-60 cm depth (Figure 4.9). Higher Cd concentrations at 0-30 cm compared to what was obtained in soils from 30-60 cm and 60-90 cm depths. The differences between soil depths were insignificant ($P > 0.05$). At a depth of 0-30 cm, the concentration of Cd was higher at the landfill at sites (1, 4, 6, 7) except for sites (2, 3, 8, 9, 10 11, 12) where the concentration was the same as the control site. The low concentration of Cd in all sites at the landfill indicates that little or no cadmium containing materials were disposed at the landfill. Cadmium is a very toxic metal that is commonly found in industrial waste including nickel-cadmium batteries and high concentrations of Cd in plants negatively affects photosynthesis and transpiration in plant.

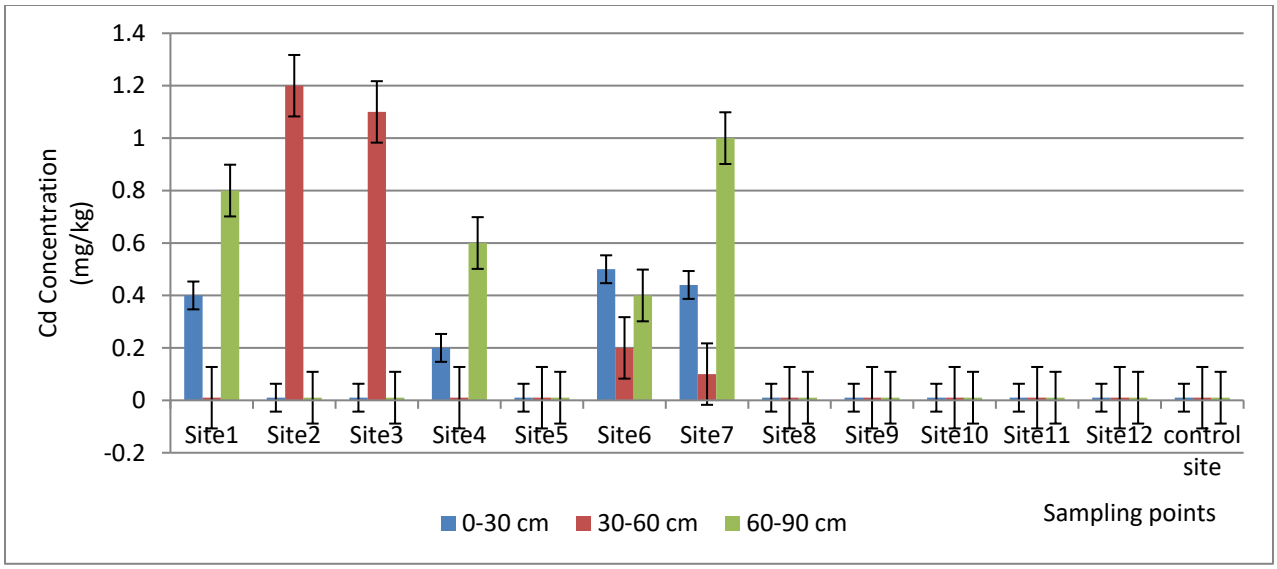


Figure 4.9: Cd concentration in soil at 0-30 cm, and at depths of 30-60 cm and 60-90 cm at the landfill

The mean Cr content of soils at the landfill was significantly higher at sites 5, 6, 7, 8, 9, 10, 11, 12 ($P<0.05$) than what was obtained at the control site (Table 4. 6). Highest concentration of Cr in the soils from the closed Lumberstewart landfill (Figure 4.10) was reported at site 6 averaging 75.0 mg/kg whereas the lowest concentration was observed in soils from site 4 averaging 23.8 mg/kg. Chromium concentrations in soils at depths of 0-30 cm ranged between 3.19 mg/kg (site 11, 12) to 39.6 mg/kg (site 6). At a depth of 0-30 cm, Cr values were lower than the Cr values at a depth of 30-60 cm ranging from 13.0 mg/kg (sites 2, 4) to 44.0 mg/kg (site 5). Cr values at a depth of 60-90 cm ranged from 13.0 mg/kg (site 1) to 48.3 mg/kg which is higher than the soils from 0-30 cm and soils at a depth of 30-60 cm. The differences between soil depths were insignificant ($P> 0.05$) for sites 5 – 12 but concentrations of Cr at sites 1 - 4 differed significantly with values obtained in soils at depths of 0 – 30 cm being significantly lower ($P<0.05$) (Figure 4.10).

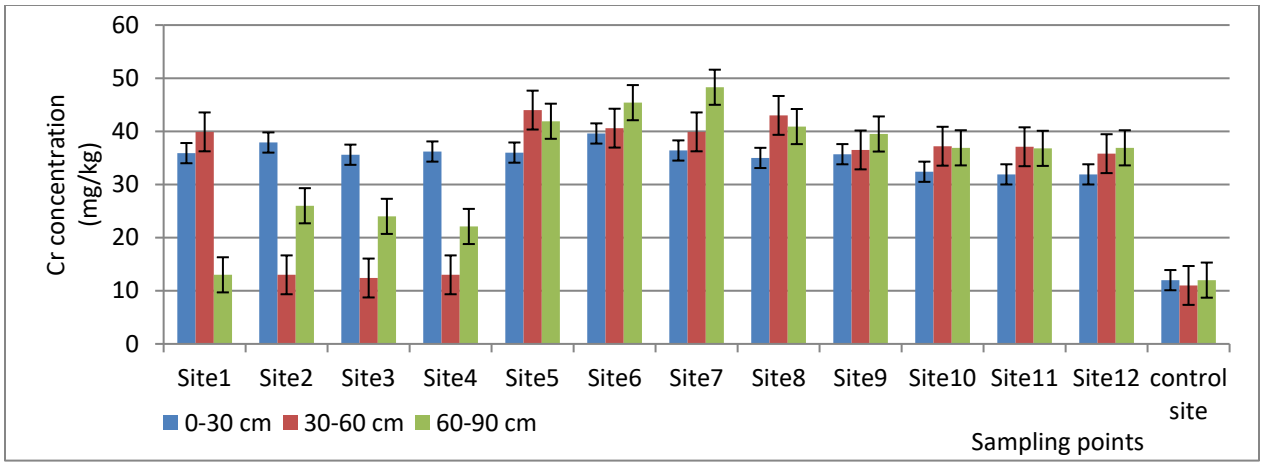


Figure 4.10: Cr concentration in soil at 0-30 cm, and at depths of 30-60 cm and 60-90 cm at the landfill.

The possible sources of waste containing Cr disposed at the landfill include ceramics, pigments, dyes and lead based paints. High levels of Cr at the landfill compared to the control site have been reported in other studies (Smith, 1996; Chrostowski et al., 1991). The toxicity of chromium depends on the form in which it is present. High concentration of Cr in the plants inhibits seed germination, reduces root growth and can alter the chloroplast and membrane (Ali et al., 2004). It has been reported that high levels of Cr in plants affects the metabolism of maize, citrullus and barley (Ali et al., 2004; Dube et al., 2003; Sharma and Pant, 1994). The values of Cr observed in the soils at the Lumberstewart landfill are higher than 6.5 mg/kg which is the permissible limits of Cr expected in soils in South Africa ((NEMA, 2008; Mtunzi et al., 2015; Durowoju et al., 2016; SABS, 1999; DNHPD, 1991). The concentration of Cr in soils at the closed landfill may be an effect in plants which are sensitive to chromium.

Copper concentration was highest at site 1, averaging 127 mg/kg whereas the least concentration was recorded in soils from site 11 with an average of 41.2 mg/kg. The amount of Cu in soils at the landfill site was almost 5 times the concentration of what was obtained in soils from the control site. The mean concentration of soil Cu at the landfill was higher than the control site (Table 4.6), the differences were insignificant ($P>0.05$). Cu concentration at a depth of 0-30 cm ranged between 49.7 mg/ kg (site 11) to 84.5 mg/kg (site 6). At a depth of 30-60 cm, Cu values were higher than the Cu values of soils at a depth of 0-30 cm with values ranging from 73.9 mg/kg in soils from site 11 to 124 mg/kg in soils from site 5 (Figure 4.11). Cu at a depth of 60-90 cm ranged between 49.6

mg/kg (site 11) to 209 mg/kg (sites 1, 6, 7) which is higher than the soils from 0-30 cm depth and soils at a depth of 30-60 cm. The differences between soil depths were insignificant (P> 0.05) (Figure 4.11).

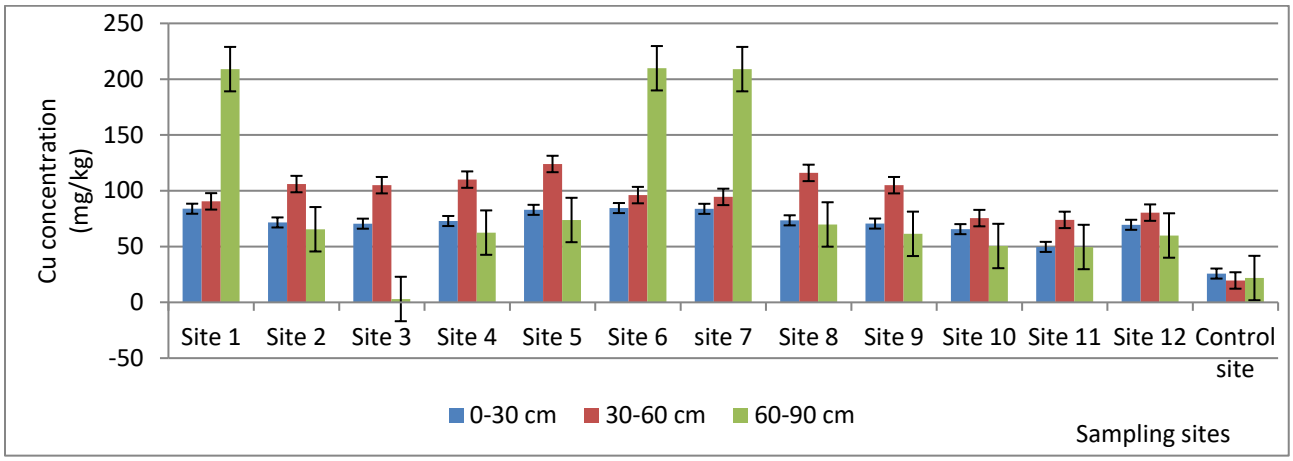


Figure 4.11: Cu concentration in soil at 0-30 cm depth, and at depths of 30-60 cm and 60-90 cm at the landfill

Cu concentrations at the landfill were higher than the permissible limits in soils (16 mg/kg) in South Africa (Mtunzi et al., 2015; Durowoju et al., 2016; SABS, 1999; DNHPD, 1991). Zhang et al. (2002) found high concentrations of Cu in a landfill in India and other studies also found that the concentrations of Cu at landfill were higher than the control sites (Pasquini and Alexander, 2004; Agyarka et al., 2010). The possible sources of Cu in soils at the landfill can be copper wires, vehicle parts, copper pipes and alloys which contain Cu. Decomposition processes in the landfill would result in acidic conditions which would contribute to the dissolution of the metal from these wastes and a high concentration in the leachate generated within the landfill. The high concentrations of Cu in the soils around the landfill could be an indication of the migration of leachate rich in Cu into the surrounding soils in the landfill.

Cu is an essential element in plant growth. It is responsible for activating enzymes in plants, assisting plant metabolism with protein catabolism and also required in the respiration process in plants. However, high concentrations of Cu in plant are detrimental in that, it could cause root damage, reduction of photosynthesis in mature leaves and disorders in plant growth (Vinit-Dunand and Badot, 2002; McBride et al., 1997). A high concentration of Cu in pigweed has been reported

to induce stress and injury to the plant (Lewis et al., 2010) and could also lead to chlorosis in jimson weed. High soil Cu concentrations could also contribute to elevated concentrations of Cu in groundwater leading to their eventual contamination. Plant and water resources around the Lumberstewart landfill therefore are at risk of Cu contamination due to leachate from the landfill.

The mean concentration of Fe in soils from the landfill was higher than what was obtained at the control site (Table 4.6) and the differences were insignifant ($P>0.05$). The amount of Fe at site 6 was about 4 times greater than the concentration of Fe in soils from the control site. Values for Fe concentration at a depth of 0-30 cm ranged from 24 351 mg/kg (site 12) to 35 592 mg/kg (site 3). These values were significantly lower than what was obtained at a depth of 30-60 cm ($p=0.00$) ($P<0.05$) with Fe concentration in soils at depths of 30 – 60 cm ranging from 29 432 mg/kg in soils from site 5 to 69 830 mg/kg in soils from site 6 (Figure 4.12). Fe concentration at a depth of 60-90 cm ranged from 24 231 mg/kg to 41 371 mg/kg which is lower than Fe concentration in soils from 30-60 cm depth ($p=0.00$) but higher than the Fe of the soils from 0-30 cm ($p=0.00$). These results indicate that Fe content in soils at the Lumberstewart landfill was exceptionally high at all sites (45690 ±17255 mg/kg). The highest concentration of Fe was observed in soils from site 6 averaging 45 690 mg/kg while the lowest concentration of Fe was observed in soils from site 9 averaging 30 872 mg/kg. High levels of Fe in the landfill samples as compared to the control site (12 080±620 mg/kg) indicate inputs into the soil from the landfill and show that waste containing iron was possibly disposed at the landfill.

High concentration of Fe in soil around dumpsites is generally common (Hoffman et al., 1991; Tripathi and Misra, 2012; Biswas et al., 2010) because Fe is contained in many waste materials that find their way into the dumpsites. Bulawayo city council, just like many other cities in the developing world does not separate industrial waste from residential and construction wastes hence there is a possibility that waste with high amount of heavy metals was disposed at the Lumberstewart landfill. High concentration of Fe in soils at the landfill could result in plants around the landfill having high concentrations of Fe (Kabata-Pendias, 1999; Batty and Younger, 2003). Fe could also leach into groundwater resulting in contamination thereof. Fe is an essential element for both plants and animals, but high concentrations of Fe could result in toxicity. According to Sinha et al. (1997), excessive Fe in tobacco and soya bean in India led to reduction

in plant photosynthesis, yields and height of the plant. Fe toxicity has also been reported in dairy cows in New Zealand (Coup and Cambell, 2012) causing dark colored foul smelling milk and watery feaces.

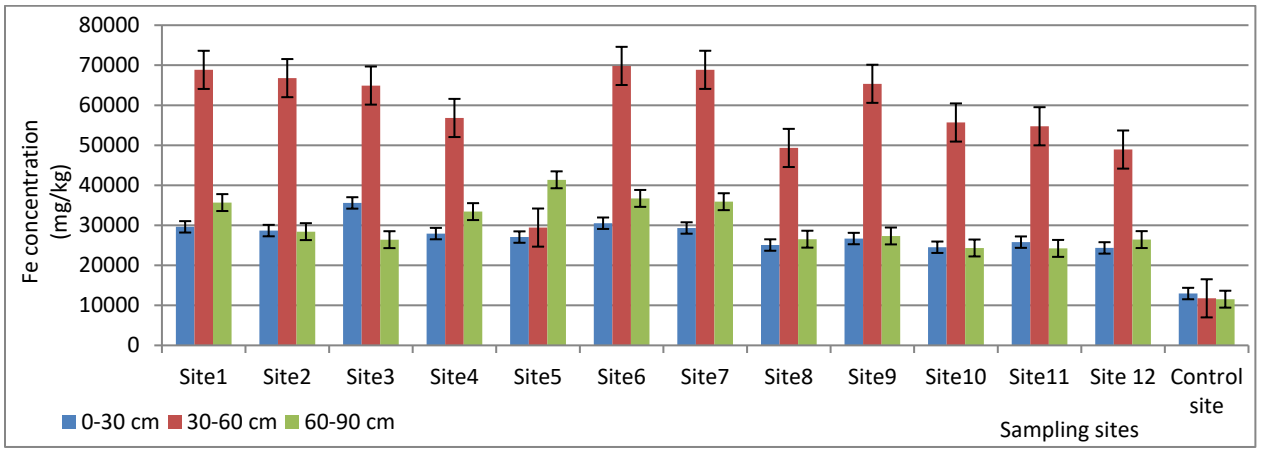


Figure 4.12: Fe concentration in soil at depths of 0-30 cm, 30-60 cm and 60-90 cm at the landfill

Mean values for Ni concentration in soils from the landfill were higher than that of the control site (Table 4.6) but the differences were insignificant (P> 0.05). At a depth of 0-30 cm, Ni concentration values ranged between 26.0 mg/kg (site 1) to 34.1 mg/kg (site 2) (Figure 4.13). Ni concentration in soils from 0-30 cm depth were lower than Ni values in soils at a depth of 30-60 with Ni values of soils at 30-60 cm ranging between 19.6 - 46.2 mg/kg (p=0.00). Ni in the soil at a depth of 60-90 cm ranged from 17.1-27.0 mg/kg which is lower than Ni values of the soils from 0-30 cm depth (p=0.01) and soils at a depth of 30-60 cm (p=0.00).The differences in Ni concentration at the different soil depths around the landfill were significant (p<0.05). Ni concentrations at the landfill were lower than the maximum permissible concentration of Ni permitted in soils (91 mg/kg) in South Africa (NEMA, 2008; Mtunzi et al., 2015; McLaughlin et al., 2000). It has been reported that the major sources of Nickel is a toxic element that is usually present in leachate generated in landfills with hazardous waste. Ni at landfills is from industries such as metal plating and electroplating, and biosolids and residue from the combustion of fossil fuels (Weggler, 2004). The presence of Ni in soils around the Lumberstewart landfill could indicate that the landfill received some industrial waste. Nickel is not an essential element and is highly toxic and carcinogenic. Its presence in soils around the landfill could present threats to plants and water resources in the area.

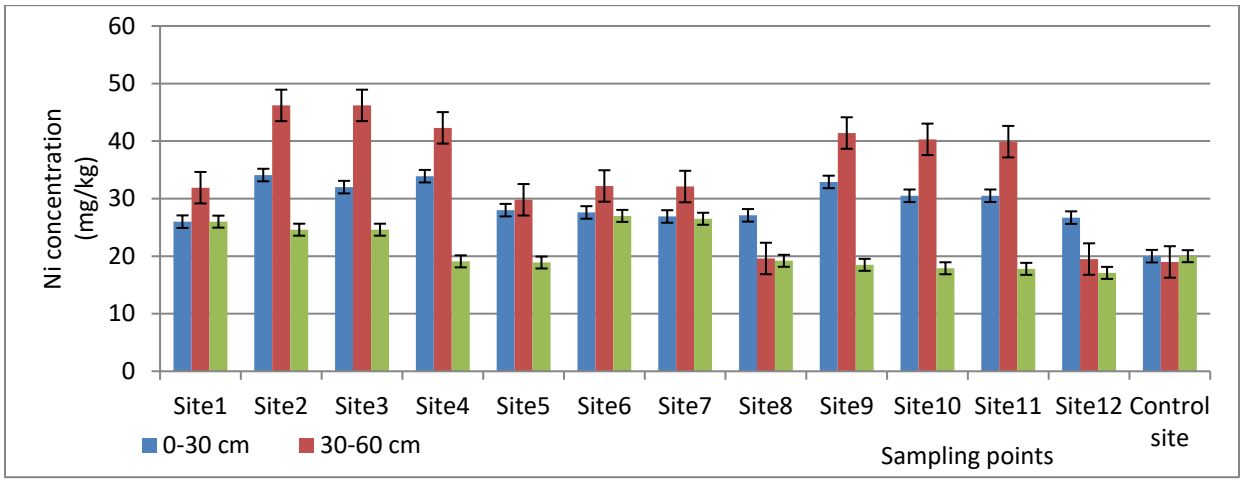


Figure 4.13: Ni concentration in soil at depths of 0-30 cm, 30-60 cm and 60-90 cm at the landfill**.**

Zn mean concentration in soils at the landfill was insignificantly higher (P>0.05) than that which was obtained at the control site (Table 4.6). Values of Zn at a depth of 0-30 cm ranged between 0.01 (site 5) to 325 mg/kg (site 4). At a depth of 0-30 cm, Zn values were lower than the Zn values in soils at a depth of 30-60 cm (p=0.00) with values of soils at 30-60 cm depth ranging from 145 mg/kg in soils from site 1 to 423 mg/kg in soils from site 5 (Fig 4.14). At a depth of 60-90 cm, Zn values ranged from 359 mg/kg to 539 mg/kg which is higher than the values from soils at depths of 0-30 cm and 30-60 cm. The differences between soil depths were significant (p =0.00) (P<0.05). The concentration of Zn at the landfill was higher than Zn concentrations expected in South Africa soils (NEMA, 2008; Mtunzi et al., 2015; Durowoju et al., 2016; SABS, 1999; DNHPD, 1991). However, the lowest concentration site had an even lesser concentration of Zn than that of control site. This could be a sign that soils in the area are generally rich in Zn. Waste from pharmaceutical facilities and textiles industries produce Zn which could find its way into landfill leachate and eventually into soils in the vicinity of the landfill. Though Zn is an essential element for plant growth, high uptake and accumulation of Zn by plants from contaminated soils could negatively affect these plants. Negative effects of Zn on the growth of pigweed have been reported by Vinit-Dunand and Badot(2002) whereas reports of the effect of Zn on the yields of jimson weed have been reported by Smith (1996).

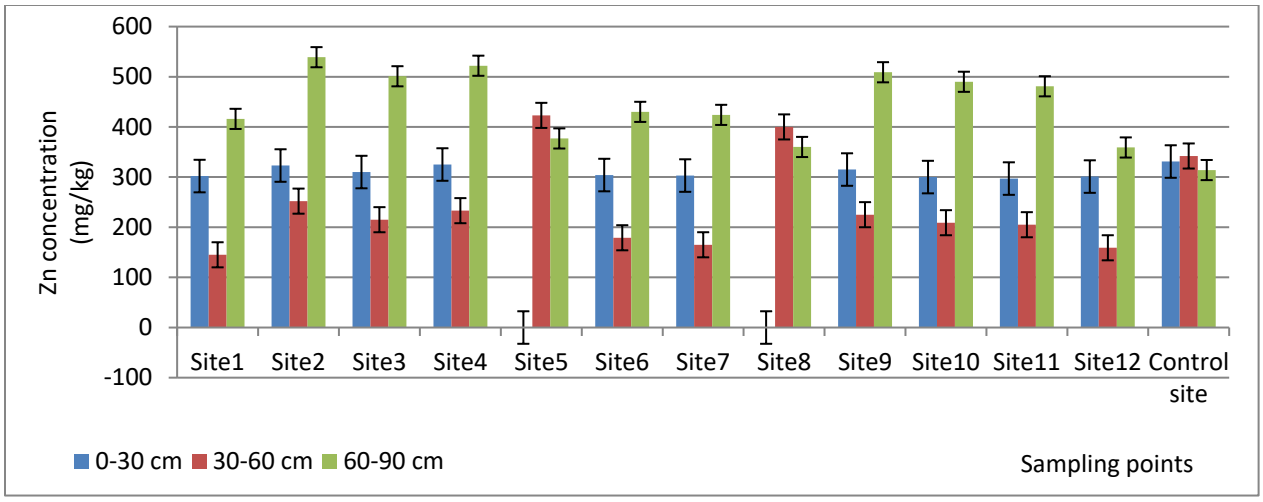


Figure 4.14: Zn concentration in soil at depths of 0-30 cm, 30-60 cm and 60-90 cm around the landfill

Results revealed that heavy metals concentration in the soils around the landfill were highest at depths of 30-60 cm. Heavy metals such as Cu, Ni and Fe were highly concentrated at 30-60 cm as compared to other depths. High concentration of heavy metals at 30-60 cm can be attributed to percolation of leachate into the subsurface carrying along with it heavy metals. This movement would have been facilitated by the coarse textured soils, low OM content and relatively low CEC. Mobility of heavy metals in the soil environment is also facilitated by the prevailing acidic soils. Metal mobility in the soil environment vary depending on soil characteristics (pH, CEC, EC, OM) as reported by authors (Mohammed and Elsayed, 2007; Hunachew and Sandip, 2011). Results reveal that there is no general trend for heavy metals analyzed with depth in soil around the Lumberstewart landfill.

4.3.2 Correlations of heavy metals with soil properties

Soil properties such as pH, soil texture, organic matter, electrical conductivity and cation exchange capacity have an effect on the movement of heavy metals in soils (Cortez and Ching, 2014; Siwela et al., 2009; Teta and Hikwa, 2017). Pearson Correlation analyses between heavy metals and soil properties at the landfill indicate very weak correlation with these soil properties (Table 4.7). There were very weak negative correlations between Cd and OM at depths of 0-30 cm, 30-60 cm and 60-90 cm. A positive correlation between Cd and CEC at a depth of 30-60 cm was recorded (Table 4.7). Cu and pH showed a positive correlation at a depth of 0-30 cm and a depth of 30-60

cm but there was a negative correlation at 60-90 cm depth and a negative correlation between Cu and OM at all depths (Table 4.8). Significant correlations were observed between Fe/pH, Fe/OM, Fe/CEC, Cu/CEC, Ni/CEC, Cd/pH and Zn/EC. Lumberstewart soil Cr also had significant relationships with soil OM and CEC as indicated in Table 4.7.

Table 4.7 Correlation between heavy metals and soil properties per depth

Metal	Soil properties											
	pH			OM			EC			CEC		
	0-30 cm	30-60 cm	60-90 cm	0-30 cm	30-60 cm	60-90 cm	0-30 cm	30-60 cm	60-90 cm	0-30 cm	30-60 cm	60-90 cm
Cd	0.03	**-0.62**	-0.35	**-0.68**	-0.16	-0.42	0.11	0.20	0.33	0.18	**0.64**	0.33
Cr	-0.47	**0.53**	0.02	**-0.73**	0.18	-0.14	**0.52**	0.18	0.37	**0.52**	**-0.55**	**-0.66**
Cu	0.33	0.02	-0.03	**-0.65**	-0.17	**-0.73**	0.49	**0.62**	0.08	0.37	0.34	0.30
Fe	0.01	**-0.53**	0.22	**-0.54**	**-0.58**	-0.37	0.05	-0.20	0.19	**0.74**	0.44	0.28
Ni	0.08	-0.14	-0.17	0.32	-0.18	**-0.88**	-0.29	-0.29	0.19	0.26	**0.50**	**0.63**
Zn	0.01	0.34	-0.17	-0.03	0.26	0.10	**-0.58**	**0.67**	-0.22	0.21	-0.09	-0.13

Among all the soil properties, OM and CEC had the most significant relationships with soil heavy metals. No clear pattern in correlation between heavy metals concentration and soil properties were observed with depth. The randomness of the correlations could be attributed to anthropogenic inputs into the soils.

4.3.3 Heavy metal enrichment in soils around the closed Lumberstewart landfill

Soil heavy metal enrichment factor (EF) around the Lumberstewart Landfill at each depth was calculated and results are presented in Table 4.8. At a depth of 0-30 cm, Ni and Zn recorded values of EF < 2 which indicate minimal enrichment, Cr, Cu and Fe recorded moderate enrichment (2 < EF < 5) and Cd had an EF value of 13.5 which shows a significant enrichment of Cd around the landfill. The 30-60 cm depth was characterized by high EF values especially for Cd and Fe (Table 4.8). At a depth of 60-90 cm, Cd and Cu had the highest EF values. Though Cd concentrations in the soils were very low, values for EF indicate significant additions of Cd into the soils at the landfill. Values for EF around the soil at different depths indicate that enrichment of heavy metals

increased with depth at the landfill. This could be an indication of the migration of leachate from the closed Lumberstewart landfill downward into the surrounding environment. There are implications of this on groundwater resources around the site.

Table 4.8: Heavy metal enrichment factor in soils at different depths at the closed landfill

Heavy Metal	Enrichment Factor (EF) at different soil depths		
	0-30 cm	30-60 cm	60-90 cm
Cd	13.5	22.3	23.9
Cr	2.90	3.00	2.90
Cu	2.80	4.70	4.50
Fe	2.20	5.00	2.60
Ni	1.50	1.80	1.10
Zn	0.80	0.70	1.40

4.4 Extent of Heavy Metal Contamination in Soils around the Landfill

Values for CF and PLI of the heavy metals in the soils in the study area are shown in Tables (4.9, 4.10, and 4.11) below. Results indicate that mean heavy metal CF values in soils around the landfill ranged from low contamination (CF<1) for Zn to moderate contamination (1 < CF< 3) for Cu Fe and Cr. These values highlight the level of contamination of the soils with heavy metals around the closed Lumberstewart landfill. Mean values for PLI of the heavy metals around the site indicate that all but sites 5 and 8 were contaminated.

Table 4.9: Contamination factor and pollution load index for metals at 0-30 cm depth at Lumberstewart closed landfill

Location	Contamination Factor (CF)						PLI
	Cu	Fe	Zn	Cd	Cr	Ni	
Site 1	3.25	2.29	0.91	40.0	3.00	1.30	32.5
Site 2	2.78	2.21	0.98	1.00	3.16	1.70	5.69
Site 3	2.73	2.75	0.94	1.00	2.97	1.60	5.79
Site 4	2.83	2.16	0.98	20.0	3.02	1.70	24.8
Site 5	3.21	2.09	0.00	1.00	3.00	1.40	0.00
Site 6	3.28	2.36	0.92	50.0	3.30	1.40	35.1
Site 7	3.25	2.27	0.92	44.0	3.03	1.30	36.6
Site 8	2.85	1.94	0.00	1.00	2.92	1.40	0.00
Site 9	2.74	2.06	0.95	1.00	2.98	1.60	5.10
Site 10	2.54	1.89	0.91	1.00	2.70	1.50	4.21
Site 11	1.93	1.99	0.90	1.00	2.66	1.50	3.71
Site 12	2.69	1.88	0.91	1.00	2.66	1.30	3.99

Note: CF<1 (low contamination); 1≤CF<3 (moderate contamination); 3≤CF<6 (considerable contamination); CF≥6 (very high contamination).

PLI>1 is polluted whereas <1 indicates no pollution (Harikumar, et at; 2009)

The most contaminated sites were 1, 6 and 7 which had the highest value for PLI. High pollution at these sites may be attributed to the kinds of waste disposed in this area of the landfill. Cd had the highest CF value (50.0) at 0-30 cm depth whereas Zn had the lowest CF at a depth of 0-30 cm indicating that soils were least polluted with Zn (Table 4.9). At a depth of 30-60 cm, Zn had the highest CF value (403) whereas Cd had the least (1.0). At a depth of 60-90 cm, Cd had the highest CF value (80) whereas Ni had the least CF (0.86). Contamination Factors were generally higher at a depth of 30-60 cm compared to 0-30 cm and 60-90 cm depths. The PLI values for the soil around the study area revealed that all sites were polluted. At 0-30 cm depth results indicated that the soils were contaminated, 10 sites had PLI > 1 except for site 5 and 8. PLI was greater than 1 at all 12 sites at a depth of 30-60 cm.

Table 4.10: Contamination factor and pollution load index for metals in the soil at a depth of 30-60 cm

Location	Contamination Factor (CF)						PLI
	Cu	Fe	Zn	Cd	Cr	Ni	
Site 1	4.62	5.86	145	1.00	3.63	1.68	154
Site 2	5.41	5.68	252	120	1.18	2.43	1632
Site 3	5.39	5.52	215	110	1.13	2.43	1390
Site 4	5.64	4.83	233	1.00	1.18	2.23	129
Site 5	6.33	2.50	423	1.00	4.00	1.57	205
Site 6	4.90	5.94	179	20.0	3.69	1.69	806
Site 7	4.82	5.86	165	10.0	3.63	1.69	534
Site 8	5.92	4.20	400	1.00	3.91	1.03	200
Site 9	5.40	5.56	223	1.00	3.32	2.18	220
Site 10	3.82	4.74	209	1.00	3.38	2.12	164
Site 11	3.77	4.66	205	1.00	3.37	2.10	159
Site 12	4.10	4.16	159	1.00	3.25	1.03	95.3

Note: CF<1 (low contamination); 1≤CF<3 (moderate contamination); 3≤CF<6 (considerable contamination); CF≥6 (very high contamination).

PLI>1 is polluted whereas <1 indicates no pollution (Harikumar, et at; 2009)

Highest PLI was observed at site 2 (1632) and lowest at site 12 (95.3) at depths of 0 – 30 cm. Pollution Load Index values were greater at a depth of 30-60 cm as compared to depths and of 0-30 cm and 60-90 cm indicating high pollution at this depth. These results further enhance the results obtained for heavy metal enrichment at different depths which indicated that the most heavy metal enrichment occurred at a depth of 30-60 cm around the landfill.

Table 4.11: Contamination factor and pollution load index for metals in the soil at 60-90 cm depth, Lumberstewart closed landfill

Location	Contamination Factor (CF)						PLI
	Cu	Fe	Zn	Cd	Cr	Ni	
Site 1	9.60	3.09	1.32	80.0	1.08	1.30	66.3
Site 2	3.00	2.46	1.72	1.00	2.17	1.23	5.82
Site 3	2.71	2.29	1.60	1.00	2.00	1.23	4.94
Site 4	2.87	2.90	1.66	60.0	1.84	0.96	38.3
Site 5	3.39	3.59	1.20	1.00	3.49	0.95	7.00
Site 6	9.62	3.18	1.37	40.0	3.78	1.35	92.5
Site 7	9.61	3.11	1.35	100	4.03	1.33	147.1
Site 8	3.20	2.30	1.15	1.00	3.41	0.96	5.26
Site 9	2.82	2.37	1.62	1.00	3.29	0.93	5.76
Site 10	2.32	2.11	1.56	1.00	3.08	0.90	4.60
Site 11	2.28	2.10	1.53	1.00	3.07	0.89	4.47
Site 12	2.75	2.29	1.14	1.00	3.08	0.86	4.36

Note: CF<1 (low contamination); 1≤CF<3 (moderate contamination); 3≤CF<6 (considerable contamination); CF≥6 (very high contamination).

PLI>1 is polluted whereas <1 indicates no pollution (Harikumar, et al; 2009)

4.5 Suitability of Soils at the Closed Lumberstewart Landfill for Various Uses

Guidelines for the maximum permissible heavy metals concentrations in agricultural soil (mg/kg) are shown below in Table 4.12. Results from this study clearly indicate that the level of heavy metal (Cu, Fe and Zn) contamination in the soil was high which may pose some serious concerns for the use of the soil to grow plants. The presence of heavy metals in the soil indicates that the soil at the landfill is being significantly affected by the landfill. Analysis above reveals that soils are highly polluted and not suitable for agricultural activities especially root crops such as carrots and groundnuts.

Table 4.12: Maximum permissible values of heavy metals in arable soils in South Africa

Heavy Metal	**Permissible value of metal in the arable soils in South Africa (mg/kg)**	**Mean Heavy metal concentration in soils around landfill (mg/kg)**
Cd	2	0.20
Cr	80	34.1
Cu	6.6	89.9
Fe	NA	38930
Ni	50	28.7
Zn	46.5	313

(*Mtunzi, et al., 2015; Durowoju, et al., 2016; SABS, 1999; DNHPD, 1991)*

The study area may also not be used as a grazing land due to high accumulation of Cu and Zn. High levels of these heavy metals in animals can cause colon cancer in animals (Agrawal, et al., 2017). According to Longhurst et al. (2004), in New Zealand and Australia it has been reported that Cd and Cu accumulation in the offal of grazing animals affect human consumption. The area may be used for recreational activities such as parks, golf courses.

4.6 Concentration of Heavy Metals in Plants

The concentration of heavy metals in pigweed (*Amaranthus*) and jimson weed (*Datura stramonium*) growing in the vicinity of the closed Lumberstewart landfill are presented in Table 4.13. Heavy metals (Cd, Cr, Cu, Fe, Ni and Zn) were present in all plant samples. Pigweed had the highest level of metal concentration in all sites as compared to jimson weed (Table 4.13). Mean concentration of heavy metals in both plants were in the order Fe>Zn>Cu>Cr>Ni>Cd. Student T-test showed that there was no significant difference between Cd concentration in jimson weed and Cd concentration in pigweed ($P=0.38$) ($P>0.05$) but there were significant differences in the mean concentration of Cd in pigweed and jimson weed from the control site and those from the landfill site ($P < 0.05$) (Table 4.13).

Table 4.13: Mean concentration of heavy metals in jimson weed and pigweed

Sites	Cd	Cu	Cr	Fe	Ni	Zn
Site 1	0.93±0.03	10.5±2.03	3.50±0.50	316±34.5	2.04±0.54	97.9±11.2
Site 2	0.03±0.02	32.7±2.70	4.03±0.50	890±11.5	0.51±0.50	48.9±11.7
Site 3	0.93±0.03	16.0±0.50	3.52±0.51	528±28.0	0.53±0.52	99.5±8.55
Site 4	2.25±0.10	12.9±1.40	2.30±0.01	328±27.0	2.35±0.45	78.3±17.4
Site 5	1.25±0.06	12.7±0.45	0.51±0.50	741±41.0	0.54±0.53	80.2±24.9
Site 6	2.93±0.03	31.2±2.25	2.78±0.48	881±26.0	2.03±0.53	110±18.8
Site 7	1.25±0.05	27.1±1.50	1.90±0.41	770±19.5	1.60±0.51	42.8±2.95
Site 8	1.20±0.05	22.0±0.50	1.78±0.33	637±2.00	1.83±0.52	66.9±7.00
Site 9	1.24±0.05	24.0±1.60	2.01±0.50	698±7.50	1.93±0.51	79.8±16.7
Site 10	1.11±0.06	20.9±0.40	1.88±0.51	660±41.5	1.82±0.48	80.2±4.35
Site 11	1.09±0.06	21.9±1.50	1.86±0.51	653±42.5	1.83±0.53	66.9±2.05
Site 12	2.14±0.05	26.8±1.15	1.85±0.49	635±30.0	1.76±0.54	52.1±2.45
Control site	0.01±0.01	3.25±1.25	0.01±0.01	334±279	1.24±0.4	11.2±0.55

The concentration of Cd in jimson weed ranged from 0.01 mg/kg to 2.90 mg/kg in whereas in pigweed, Cd values ranged from 0.05 mg/kg to 2.95 mg/kg (Table 4.13). Cd concentrations in both plants were highest in plants from site 6. The lowest concentration of Cd in both plants was observed in plants from site 2. Cadmium values obtained from the plants indicate that pigweed had a higher concentration of Cd as compared to jimson weed in all 12 sites. (Table 4.13). The differences between Cd concentration in jimson weed and Cd concentration in pigweed were insignificant ($p= 0.79$) ($P> 0.05$) (Figure 4.15).

The concentration of Cr in the plants ranged from 0.01 mg/kg to 3.53 mg/kg in jimson weed whereas in pigweed Cr concentration ranged from 1.00- 4.52 mg/kg. This indicates a higher concentration of Cr in jimson weed compared to pigweed. Cr concentrations in both plants were highest in plants from sites 2 compared to the other sites. The lowest concentration of Cr in plants was observed in plants from site 5. The mean Cr concentration in the plants from the landfill was significantly higher (sites 1, 2, 3) than the mean values recorded from the control site ($P<0.05$)

(Table 4.13). The differences between Cr concentration in jimson weed and pigweed were insignificant (p = 0.08) (P> 0.05) (Figure 4.16).

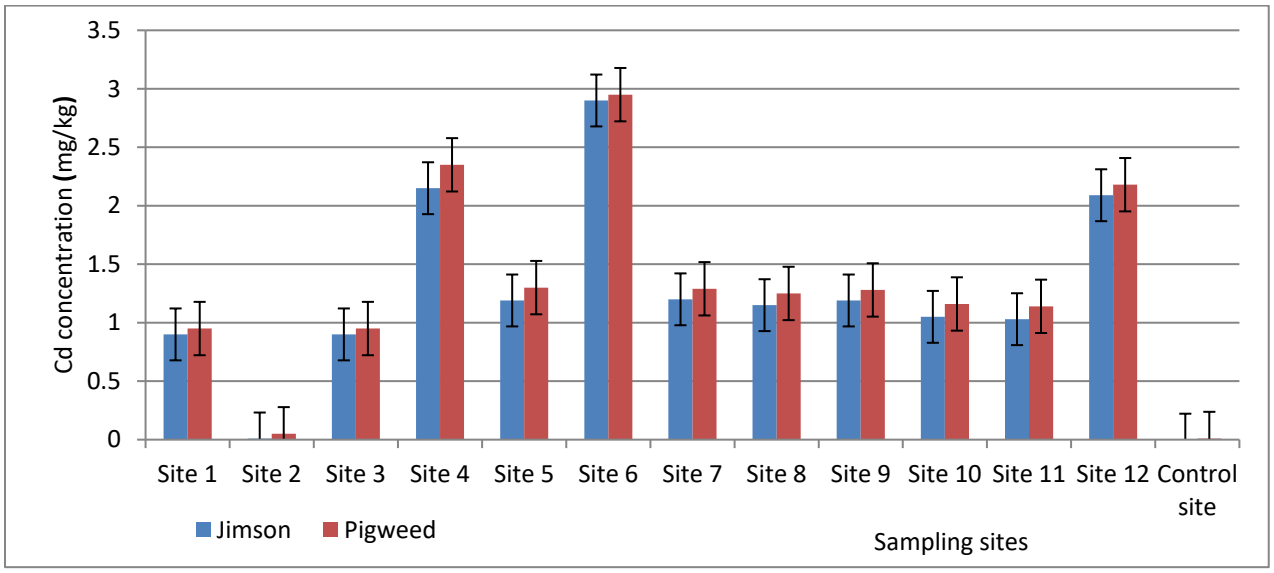


Figure 4.15: Cd concentration in jimson weed and pigweed at the landfill

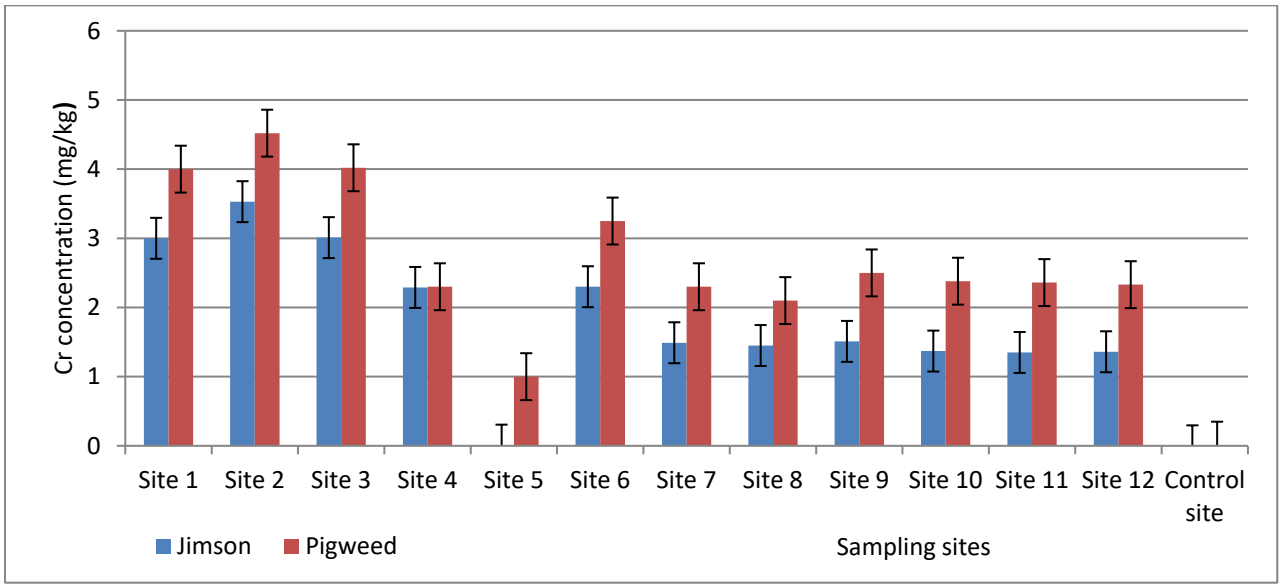


Figure 4. 16: Cr concentration in jimson weed and pigweed at the landfill

The concentration of Cu in jimson weed and pigweed on most sites ranged between 8.44-30.0 mg/kg in jimson weed whereas in pig weed ranged between 12.5-35.4 mg/kg (Figure 4.13). These

values also indicate a higher concentration of Cu in jimson weed compared to pigweed. The pattern of Cu concentration in plants from the different sites around the landfill differed from that of Cr with Cu plant concentrations from the different sites following the order Site 2 > Site 6 > Site 7 > Site 12 > Site 9 > Site 8 > site 11 > Site 10 > Site 3 > site 5 = Site 4 > Site 1 (Figure 4.17).

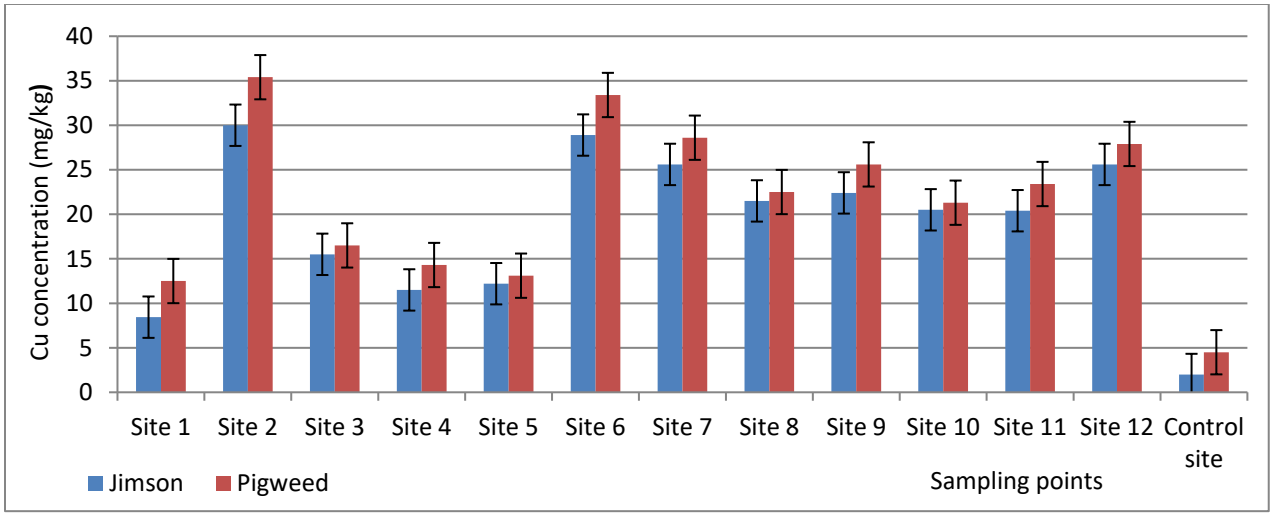


Figure 4. 17: Cu concentration in jimson weed and pigweed at the landfill

Copper concentration in jimson weed and pigweed at the landfill sites were significantly higher (sites 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12) than Cu values obtained from the control site (Table 4.13). The differences between Cu values in jimson weed and Cu values in pigweed at the landfill were insignificant (p=0.44).

The amount of Fe in both plants were ranging between 281 and 855 mg/kg in jimson weed and 350 and 907 mg/kg in pig weed. Results of Fe concentration in plants from the different sites showed that Fe concentration was the highest in both jimson weed and pig weed. Site 2 recorded the highest concentration of Fe with values of 878 g/kg in jimson weed whereas site 6 had the highest concentration of Fe in pigweed with values of 907 mg/kg. Results obtained reveal that the differences of Fe in jimson weed and pigweed at the landfill were insignificant (Figure 4.18). These results are not unexpected considering the high concentrations of Fe in the soils on which the plants are growing. The mean values for Fe in jimson weed and pigweed at the landfill site were higher than mean values of Fe in the same plants at the control site, the differences are insignificant (p=0.28) (P> 0.05). (Table 4.13).

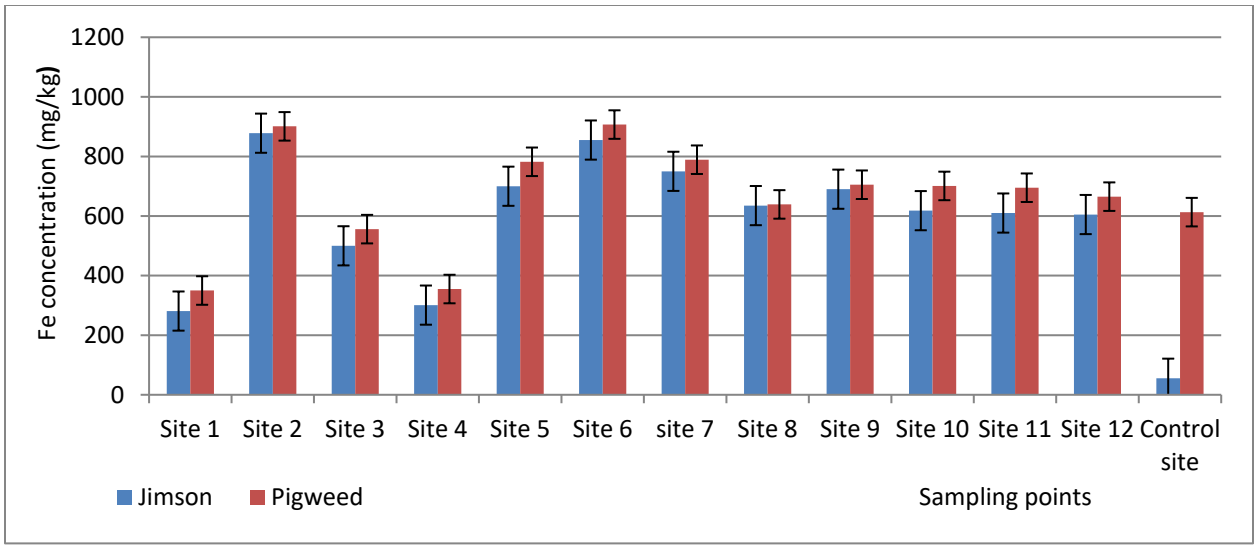


Figure 4.18: Fe concentration in jimson weed and pigweed at the landfill

The concentration of Ni in the plants ranged from 0.01 mg/kg to 1.90 mg/kg in jimson weed whereas in pigweed, Ni concentration ranged from 1.00 to 2.80mg/kg. Nickel concentrations in both plants were highest in plants from site 4. The lowest concentrations of Ni in plants were observed in plants from sites 2, 3 and 5. The mean concentration of Ni in jimson weed and pigweed at the landfill were significantly higher than Ni concentration in the same plants at the control site (Table 4.13) except for sites 2, 3 and 5. The differences between Ni in jimson weed and pigweed were significant ($P<0.05$) ($p= 0.01$). (Figure 4.19). This shows a higher concentration of Ni in pigweed compared to jimson weed.

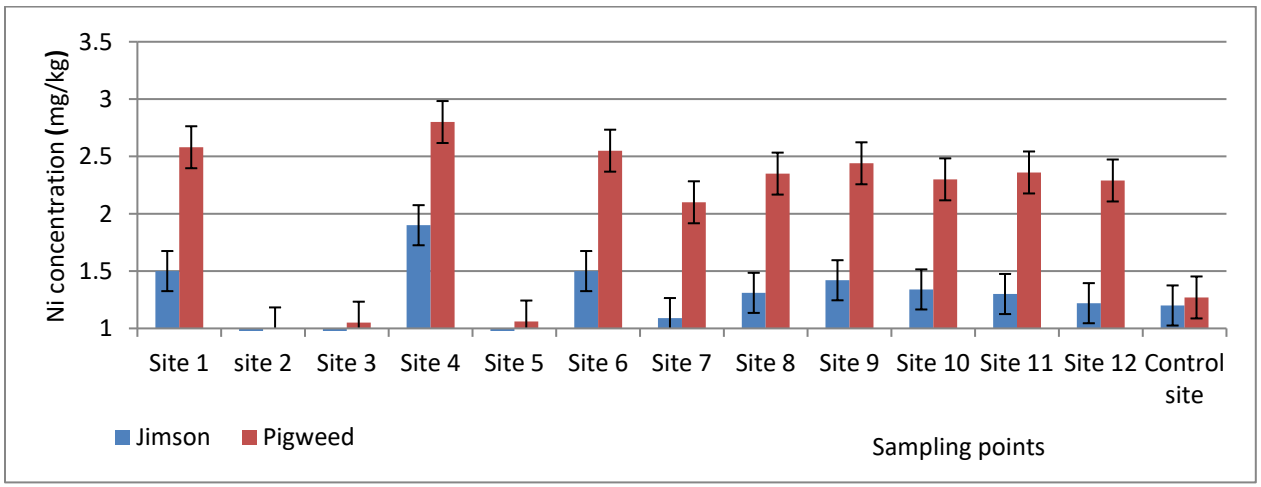


Figure 4. 19: Ni concentration in jimson weed and pigweed at the landfill

Zinc concentration in plants ranged from 37.2 mg/kg to 91.5 mg/kg in jimson weed whereas in pigweed Zn ranged from 45.7 mg/kg to 129 mg/kg. Pigweed had higher concentration of Zn as compared to jimson weed. Differences between plants jimson weed and pigweed at the landfill were insignificant ($p= 0.08$) ($p> 0.05$). (Figure 4.20). Zn concentrations in both plants were highest in plants from site 6. The lowest concentration of Zn in jimson weed was observed in plants from site 2 whereas in pigweed it was observed in plants from site 7. The mean concentration of Zn in plants at the landfill were higher than the plants from the control site (Table 4.13), the differences were significant ($p<0.05$) at sites 1, 5 and 6.

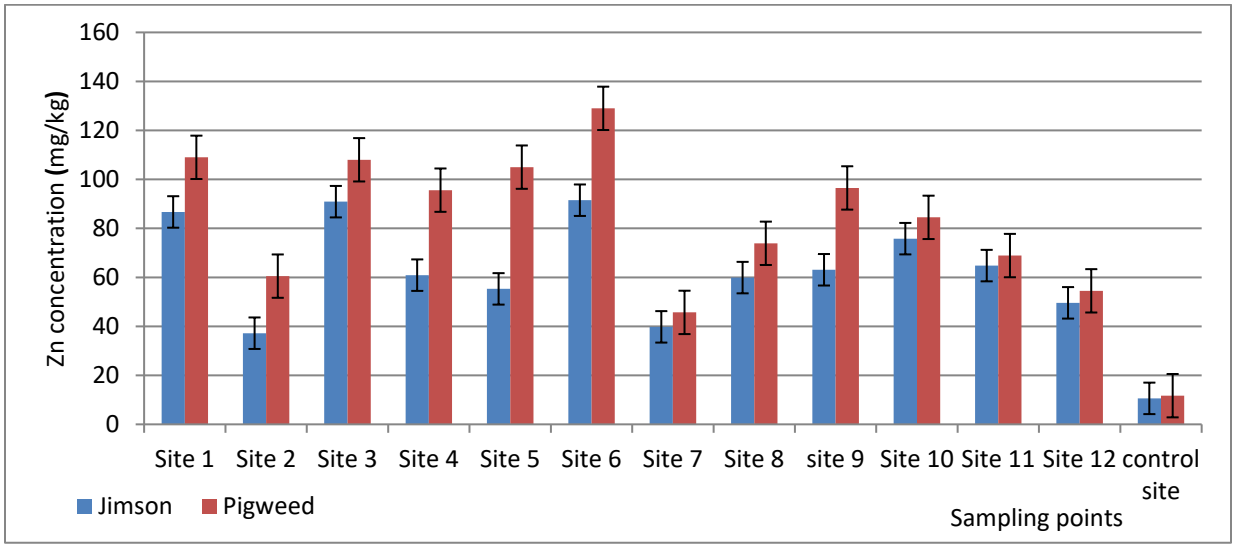


Figure 4. 20: Zn concentration in jimson weed and pigweed at the landfill

4.7 Heavy Metals Bioaccumulation by Jimson Weed and Pigweed growing on Soils at the Closed Lumberstewart Landfill

Transfer factor (TF) refers to the ratio of the concentration of heavy metal in a plant to the concentration of heavy metal in soil. Transfer Factor shows the amount of heavy metals in the soil that ends up in the plant (Harrison and Chirgawi, 1989; Smith and Doran, 1996). Cadmium recorded the highest value of TF for pigweed and jimson weed (0.95 and 0.90) respectively at site 1 (Table 4.14). This might be due to high mobility of Cd and low retention of the metal in the soil compared to other metals under study (Lokeshwan and Chandrappa, 2006).

Table 4.14: Transfer factors of heavy metals in jimson weed and pig weed

Location	Plant type	Metal Concentrations (mg/kg) in two plants studied					
		Cd	Cr	Cu	Fe	Ni	Zn
Site 1	Jimson weed	0.90	0.10	0.07	0.006	0.05	0.30
	Pig weed	0.95	0.14	0.10	0.007	0.09	0.38
Site 2	Jimson weed	0.25	0.05	0.37	0.022	0.01	0.10
	Pig weed	0.13	0.07	0.03	0.023	0.03	0.17
Site 3	Jimson weed	0.13	0.13	0.20	0.013	0.09	0.27
	Pig weed	0.17	0.17	0.44	0.014	0.03	0.32
Site 4	Jimson weed	0.27	0.10	0.14	0.007	0.06	0.17
	Pig weed	0.32	0.10	0.17	0.008	0.09	0.27
Site 5	Jimson weed	0.21	0.01	0.13	0.021	0.00	0.21
	Pig weed	0.40	0.05	0.14	0.024	0.05	0.40
Site 6	Jimson weed	0.30	0.03	0.43	0.019	0.05	0.30
	Pig weed	0.43	0.04	0.50	0.021	0.08	0.43
Site 7	Jimson weed	0.13	0.02	0.39	0.002	0.38	0.13
	Pig weed	0.15	0.03	0.43	0.003	0.07	0.15
Site 8	Jimson weed	0.24	0.04	0.25	0.018	0.06	0.24
	Pig weed	0.29	0.05	0.26	0.021	0.15	0.29
Site 9	Jimson weed	0.18	0.04	0.28	0.022	0.05	0.18
	Pig weed	0.28	0.07	0.32	0.023	0.08	0.28
Site 10	Jimson weed	0.23	0.05	0.32	0.018	0.05	0.23
	Pig weed	0.25	0.07	0.33	0.020	0.08	0.25
Site 11	Jimson weed	0.20	0.06	0.5	0.017	0.04	0.20
	Pig weed	0.21	0.07	0.57	0.020	0.09	0.21
Site 12	Jimson weed	0.18	0.04	0.37	0.018	0.06	0.18
	Pig weed	0.20	0.07	0.40	0.020	0.11	0.20

On the other hand, Ni had the lowest value of TF (0.00) in jimson weed at site 5 and also the lowest value of TF (0.003) was recorded in Fe for pigweed at site 7. There were no significant differences between the TF of heavy metals with the exception of Cd in which TF values ranged between 0.13 to 0.95. Generally the results indicate that heavy metal TF was less than one in both plant species. TF values indicate that the studied heavy metals accumulated in plants to different extent. Values

for TF were higher in plant samples having low heavy metal concentration (Cd) compared to samples with high metal concentration (Fe) which concur with the previous findings by Lokweshwan and Chandrappa (2006), Smith and Doran (1996) and Li (2009). The low TF indicate that the concentration of the studied heavy metals in the two weeds does not depend on their concentration in the soil but on the plant species.

These results further buttress the fact that total concentrations of heavy metals in soils do not necessarily mean potentially high uptake by plants growing on such soils. Soil heavy metal concentrations simply present the risk to which plants may be exposed. The two weeds currently growing on the site may have tolerance for these metals which may explain why they are thriving in this environment despite the high metal concentration in the soil. This is further justified by the low TF Values. Values for TF may also indicate that these weeds selectively take up heavy metals which may explain why despite the high concentrations of heavy metals in the soils, the TF values are low. Though this may be the case with the weeds studied, the situation may be different with food crops. Recent researches indicate that high concentrations of heavy metals such as Cu, Fe, Zn, Cr, Cd and Ni in soils have significant adverse impact on human beings, animals and plants. In countries such as Iraq, India and China it is claimed that millions of people are potentially at risk from heavy metals poisoning (Tripathi and Misra, 2012).

High concentration of heavy metals in plant samples can be attributed to the availability of the heavy metals in the soil on which the plants are growing and the ability of plant roots to uptake these metals. Sekara et al. (2005); Siwela et al. (2009) and Teta and Hikwa (2017) also reported high concentration of heavy metals in plants around a closed landfill. Results indicated that heavy metals are transmitted from soils to plants at the Lumberstewart landfill. Heavy metal concentrations in both plants are detrimental since they are above WHO permissible limits (WHO, 1996) except for Ni which was below the permissible value for plants (10.0 mg/kg) at all the sites studied around the landfill. The community might be affected through ingestion of pigweed and jimson containing heavy metals. The effects of high intake on heavy metals have been reported by Cambra et al. (1999) and Nicholls and Mal (2003). Exposure of humans using these weeds could result in death of unborn fetus and lung cancer among others (Kibra, 2008). According to Jaloon and Malik (2017), high consumption of iron may lead to DNA damage and loss of memory.

CHAPTER 5
SUMMARY OF FINDINGS, CONCLUSION AND RECOMMENDATIONS

5.1 Introduction

This chapter synthesizes the entire research under study. It provides a brief summary of findings and recommendations linked to the problem of the study and objectives of the research. The chapter also provides the conclusion of the work under investigation aiming to improve the management of a closed landfill.

5.2 Summary of Findings

The purpose of the study was to assess the impacts of a closed landfill on the heavy metal concentration in soils and plants in the vicinity of the landfill. The study sought to determine the concentration of selected heavy metals (including Cd, Cr, Cu, Ni, Fe and Zn) in soils, how heavy metals concentrations in soils around the closed landfill vary with depth, the amount of heavy metals in plants in the vicinity, whether heavy metal concentration in plants around the landfill vary with species, and how the concentration of heavy metals in soils and plants deviate from acceptable standards.

Based on the findings of the study, soils at the landfill were mainly dominated by sand, followed by clay and then silt. Sand content decreased from 0-30cm to a depth of 60-90 cm whereas clay content increased with depth. There was no difference in soil texture between landfill sites and control site. The soil pH ranged between 5.01 at 60-90 cm to 7.65 at 30-60 cm. Soil pH ranged from slightly acidic to alkaline. The pH values decreased with an increase in depth. At the control site, soil pH was alkaline compared to soils at the landfill. Soil electrical conductivity indicated inputs from the landfill but the soils cannot be described as saline. There was no significant difference in EC among sites at different sites. Results indicated that there was a significant difference between organic matter content in soils from 0-30 cm depth of the landfill and soils at a depth of 30-60 cm. Soil CEC was highest at surface and lowest at a depth of 60-90 cm.
The information from the study also indicated variation of heavy metals concentration in soil with depth. Generally, there was no pattern for accumulation of heavy metals from 0-30 cm to a depth of 60-90 cm. Cd, Fe and Ni were highly concentrated in soils at a depth of 30-60 cm whereas Cr, C and Zn were highly concentrated at soil depths of 60-90 cm. Results also indicate that heavy metals were more concentrated at soil depths of 0-30 cm. Heavy metal concentrations at the landfill site were higher than the control site. The research also revealed that the CF for heavy metals studied was above one (CF>1) showing moderate, considerable and very high contamination. All the values obtained at the landfill were greater than 1 (PLI>).

The study further found out that Cd, Cr, Cu, Fe, Ni and Zn were present in both plant species studied around the landfill. Pigweed accumulated highest concentration of heavy metals at all sites compared to jimson weed. Results indicated that correlation between heavy metal in plant and soil was mostly insignificant. However, there was a positive correlation between heavy metal (Zn and

Cr) in soil and both plants (pigweed and jimson weed). The TF was generally low at the landfill especially where the soil had higher level of heavy metals. Cadmium recorded the highest value for TF whilst Fe recorded the least TF value. The TF was generally lower than one for all the samples.

5.3 Conclusion

Soil pH was slightly acidic. The acidic pH of soils would enhance the mobility of heavy metals in the soils because most of heavy metals occur in soluble forms at acidic pH levels. EC at the landfill was low. Soil salinity was therefore very low at the landfill. The soils also contained low content of OM and CEC which could further enhance the soil's inability to adsorb heavy metals from migrating leachate.

The mean concentrations of heavy metals in the soils were higher than the control site. Fe concentrations in the soils were exceptionally high whereas Cd had the lowest concentration at the landfill. The mean concentration of heavy metals at the landfill was above permissible values of metals in the arable soils in South Africa.

Results obtained from the study indicated that there is no specific pattern for heavy metal concentrations with depth in the soils around the landfill. There were slight changes for Cd from depth to depth as compared to other heavy metals. Significant correlations were observed between heavy metals and soil properties per depth. The CF values obtained ranged from low contamination to very high contamination. The EF and CF of heavy metals increased with depth at the landfill indicating that there could be vertical migration of leachate from the closed Lumberstewart landfill.

The accumulation of heavy metals in soils and plants in the study area is of great concern since the area is used for agricultural and recreational activities. This may present risks to plants and water resources in the vicinity of the landfill.

5.4 Recommendations

The current study has offered a better understanding on the concentrations of heavy metals in soils and plants at the closed landfill. There are still some more investigations that need to be done in order to improve our knowledge with regard to handling of municipal solid waste in a manner that would not compromise the well-being of water, plants and soil resources. Results revealed that the closed landfill is still releasing many potent contaminants to the environment. Based on the results of this research the recommendations put forward include the following:

- There should be continuous groundwater monitoring at the closed landfill site as there are indications that migration of leachate from the closed landfill is ongoing.
- The municipality should dissuade the use of the soils around closed landfills to grow crops because there is a likelihood that the soils are contaminated. Where crops or vegetables are grown they should not be harvested for human or animal consumption.
- Efforts should be put in place to discourage the consumption of pigweed and jimson weeds as they could contribute to heavy metal exposure of residents in the area.

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APPENDIX 1: PERMISSION LETTER FROM THE CITY OF BULAWAYO

City of Bulawayo

All Communications
To be addressed to the
Town Clerk

Town Clerk's Office
Municipal Buildings
Fife Street
P.O.Box 591
Bulawayo

Tel: (263-9) 75011
Fax: (263-9) 69701
Email: tcdevt@citybyo.co.zw
Website: www.citybyo.co.zw
Facebook: The City of Bulawayo
Twitter:@CityofBulawayo
Call Centre: 08084700 (Econet,)
08004700 (Telone) (09) 71290

REF: JBM/MZ.74-00-00

14/12/17

Miss P. Makuleke
University of South Africa
P.O. Box 392
South Africa

Dear Mr/Mrs/Miss P. Makuleke

RE: REQUEST FOR PERMISION TO CARRY OUT RESEARCH ON COUNCIL PREMISES: AN ASSESSMENT OF THE IMPACTS OF A CLOSED LANDFILL ON SOILS AND PLANTS.

Your letter on the above matter refers.

Please be informed that Council acceded to your request to carry out research within Bulawayo City Council premises subject to the following conditions:

a) You should submit a copy of your research findings after completing the research exercise.
b) Council is to be indemnified against any accident/mishaps, which may occur during the conduct of the research.

Accordingly you may approach any of Council's Service Departments as appropriate for assistance.

Yours faithfully

TOWN CLERK

CITY OF BULAWAYO
TOWN CLERK'S DEPT.
14 DEC 2017
HUMAN RESOURCES DIVISION
P.O. BOX 591, BULAWAYO
TEL: +263 (9) 75011

APPENDIX 2: ETHICS APPROVAL

UNISA university of south africa

CAES HEALTH RESEARCH ETHICS COMMITTEE

Date: 31/01/2019

NHREC Registration # : REC-170616-051
REC Reference # : 2018/CAES/009
Name : Ms P Makuleke
Student #: 53154649

Dear Ms Makuleke

Decision: Ethics Approval Renewal after First Review from 01/02/2019 to 31/01/2020

Researcher(s): Ms P Makuleke
peacemaka@gmail.com

Supervisor (s): Prof VM Ngole-Jeme
ngolevm@unisa.ac.za; 011-471-3878

Working title of research:

An assessment of the impacts of a closed landfill on soils and plants: A case of Lumberstewart landfill near Sobukhazi, Bulawayo Zimbabwe

Qualification: MSc Environmental Management

Thank you for the submission of your progress report to the CAES Health Research Ethics Committee for the above mentioned research. Ethics approval is renewed for a one-year period. After one year the researcher is required to submit a progress report, upon which the ethics clearance may be renewed for another year.

Due date for progress report: 31 January 2020

Please note the points below for further action:

1. The researcher is cautioned to adhere to the stipulations in the permission letter from the municipality.

*The **low risk application** was **reviewed** by the CAES General Research Ethics Review Committee on 01 February 2018 in compliance with the Unisa Policy on Research Ethics and the Standard Operating Procedure on Research Ethics Risk Assessment.*

university of south africa
Preller Street, Muckleneuk Ridge, City of Tshwane
PO Box 392, UNISA, 0003 South Africa
Telephone +27 12 429 3111 Facsimile +27 12 429 4150
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APPENDIX 3: RAW AND UNPROCESSED DATA OF THE SAMPLES

Soil texture at Lumberstewart closed landfill

Site	Weight percent (%)								
	0-30 cm			30-60 cm			60-90 cm		
	Sand	Silt	Clay	Sand	Silt	Clay	Sand	Silt	Clay
Site 1	67.4	14.5	18.1	63.90	16.4	19.7	61.6	15.2	23.2
Site 2	71.5	13.5	15.0	67.5	15.4	17.1	66.9	14.5	18.6
Site 3	65.7	14.9	19.4	65.6	16.9	17.5	62.5	17.1	20.4
Site 4	62.9	16.8	20.3	64.5	13.9	21.6	65.6	13.1	21.3
Site 5	74.5	11.3	14.2	72.6	12.2	15.2	71.5	12.0	16.5
Site 6	74.2	10.5	15.3	75.9	9.5	14.6	76.5	10.1	13.4
Site 7	69.0	14.5	16.5	70.5	9.4	20.1	71.6	10.1	18.3
Site 8	74.4	9.3	16.3	76.2	10.3	13.5	76.4	10.4	13.2
Site 9	76.3	10.6	13.1	74.9	10.9	14.2	76.4	10.6	13.0
Site 10	74.9	10.6	14.5	71.8	10.6	17.6	70.5	9.4	20.1
Site 11	72.4	11.1	16.5	69.5	10.9	19.6	68.5	9.4	22.1
Site 12	68.5	16.4	13.1	62.6	9.7	12.9	61.7	10.5	11.5
Control site	78.4	10.2	11.4	81.4	6.7	11.9	79.8	7.7	12.5

PROPERTIES OF SOILS AROUND THE LANDFILL

	pH			CEC			EC			OM		
SAMPLE ID	0-30cm	30-60 cm	60-90 cm	0- 30cm	30-60 cm	60-90 cm	0-30 cm	30-60 cm	60-90 cm	0-30 cm	30-60 cm	60-90 cm
1	6.1	6.9	6.06	15.68	14.35	14.04	0.52	0.65	0.61	1.5	1.3	1.44
2	6.11	6.09	6.04	14.6	13.5	11.15	0.55	1.2	1.3	1.46	1.21	1.35
3	6.0	6.02	6	15.6	14	13.9	0.59	0.78	0.79	1.78	1	1.52
4	6	6.9	5.85	10.5	10.45	10.1	0.43	0.53	0.55	2.3	1.9	2.25
5	6.4	7.65	6.47	9.1	8.88	8.68	1.04	1.18	1.25	1.91	1.69	1.85
6	6.55	6.85	6.2	9	8.92	8.8	1.18	1.29	1.61	1	0.8	0.98
7	5.5	6.64	5.4	8.19	8.54	8.75	0.5	0.5	0.57	1.48	1.05	1.1
8	5.76	6.6	6.21	7.66	7.61	7.45	1.01	1.56	1.6	1.92	1.82	1.65
9	6.42	6.4	6.3	8.1	8.05	7.96	0.9	0.95	1	2.5	1.95	2.16
10	6.34	7.16	6.12	7.8	8.9	9.5	0.4	0.62	0.74	2	1.95	1.65
11	5.5	7.07	5.01	5.3	5	4.3	0.39	0.51	0.56	2.35	1.99	2.2
12	6.25	6.18	6.6	5.45	5.01	4.45	0.56	0.64	0.6	2.4	2	2.15
CONTROL SITE	7.6	7.25	7	6.4	5.4	3.5	0.3	0.26	0.15	1.5	1.2	1

CONCENTRATION OF HEAVY METALS IN SOILS AROUND THE LUMBERSTEWART CLOSED LANDFILL

	Cd			Cr			Cu		
SAMPLE ID	**0-30 cm**	**30-60 cm**	**60-90 cm**	**0-30 cm**	**30-60 cm**	**60-90 cm**	**0-30 cm**	**30-60 cm**	**60-90 cm**
1	0.40	0.01	0.80	35.9	39.9	13.0	83.9	90.5	209
2	0.01	1.20	0.01	37.9	13.0	26.0	71.6	106	65.5
3	0.00	1.10	0.01	35.6	12.4	24.0	70.5	105	59
4	0.20	0.01	0.60	36.2	13.0	22.1	72.9	110	62.5
5	0.01	0.01	0.01	36.0	44.0	41.9	82.9	124	73.8
6	0.50	0.20	0.40	39.6	40.6	45.4	84.5	96.1	209
7	0.44	0.10	1.00	36.4	39.9	48.3	83.8	94.5	209
8	0.01	0.01	0.01	35.0	43.0	40.9	73.5	116	69.8
9	0.01	0.01	0.01	35.7	36.5	39.5	70.6	105	61.4
10	0.01	0.01	0.01	32.4	37.2	36.9	65.6	75.5	50.5
11	0.01	0.01	0.01	31.9	37.1	36.8	49.7	73.9	49.6
12	0.01	0.01	0.01	31.9	35.8	36.9	69.5	80.4	59.9
CONTROL SITE	0.01	0.01	0.01	12.0	11.0	12.0	25.8	19.6	21.8

	Fe			Ni			Zn		
SAMPLE ID	**0- 30 cm**	**30-60 cm**	**60-90 cm**	**0-30 cm**	**30-60 cm**	**60-90 cm**	**0-30 cm**	**30-60 cm**	**60-90 cm**
1	29621	68842	35684	26	31.9	26	302	145	416
\Q2	28682	66782	28417	34.1	46.2	24.6	323	252	539
3	35592.0	64921	26415	32	46.2	24.6	310	215	501
4	27935	56821	33421	33.9	42.3	19.1	325	233	522
5	27062	29432	41371	28	29.8	18.9	0.01	423	377
6	30521	69830	36721	27.6	32.2	27	304	179	430
7	29331	68851	35900	26.9	32.1	26.5	303	165	424
8	25071	49332	26533	27.1	19.6	19.2	0.01	400	360
9	26691	65361	27337	32.9	41.4	18.5	315	223	509
10	24521	55690	24337	30.5	40.3	17.9	300	209	490
11	25791	54751	24231	30.5	39.9	17.8	297	205	481
12	24351	48931	26431	26.7	19.5	17.1	301	159	359
CONTROL SITE	12949	11753	11539	20	19	20	331	342	314